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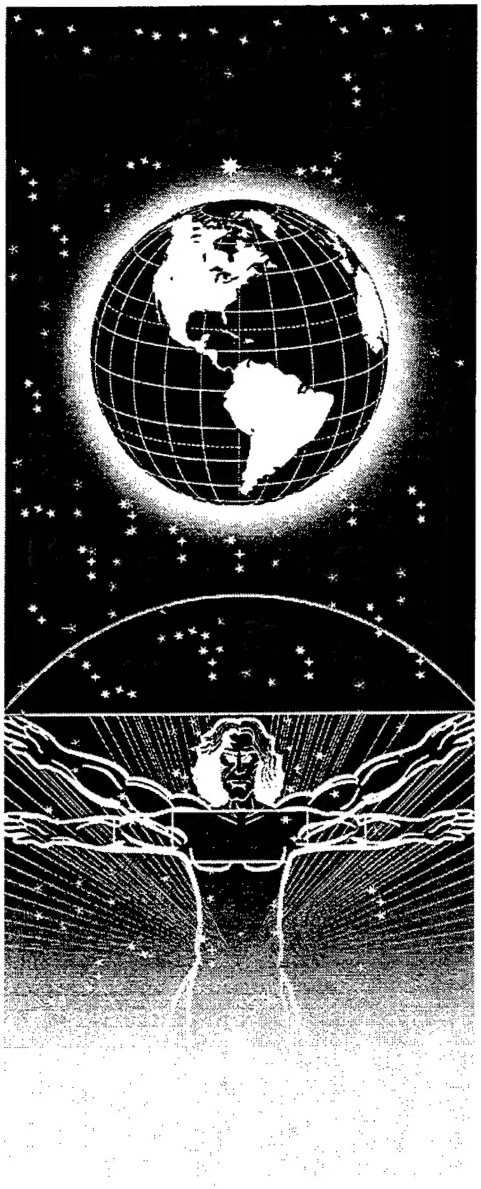

UNITED STATES AIR FORCE ARMSTRONG LABORATORY

Review of Exposure Assessment Guidelines

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This report is a review of the exposure assessment guidelines recommended by the U.S. Environmental Protection Agency, other federal and state agencies, and nongovernmental research groups. Emphasis is placed on describing the general concepts used in exposure assessments and in identifying the population-, site- and chemical-specific exposure parameters that are used or recommended by these groups. Also included is an overview of the recommendations made by EPA's Science Advisory Board, and the National Research Council concerning current exposure assessment methodologies. The primary objective of this report is to identify those steps in the exposure assessment process for which there might be alternative approaches which may lead to an overall improvement in the cost efficiency and quality of the exposure assessments conducted on federal sites.

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TABLE OF CONTENTS

TABLE OF CONTENTS (Continued)

TABLE OF CONTENTS (Continued)

LIST OF TABLES

PREFACE

This document was produced under the auspices of the Environmental Risk Assessment Program (ERAP), which has its genesis in the DOD/DOE Strategic Environmental Research and Developmental Program (SERDP) that was established through Public Law 101-510 (10 United States Code 2901-2904). ERAP was established as a cooperative effort of DOD, DOE, and EPA to improve health and ecological risk assessments and to foster consistency in risk assessments across federal agencies. The program has three working groups chartered under its mission which are the Materials/Chemicals Risk Assessment (MCRA) Working Group, Human Risk Assessment Methodology (HRAM) Working Group, and the Ecological Risk Assessment Methodology (ERAM) Working Group. The program also has an Advisory and Coordinating Committee (ACC) that oversees the program and the working group's activities.

This document is a consensus product of the HRAM Working Group which includes Maj. B. Larcom (chairperson) from the U.S. Air Force; Ms. V. Hauschild from the U.S. Army; Ms. A. Lunsford from the U.S. Navy; Dr. Richard Hertzberg and Ms. C. Scott from the U.S. Environmental Protection Agency; Dr. M. Frazer from the Department of Energy, and Dr. K. Davidson, Dr. R. Ross and Dr. P. Lu of the Oak Ridge National Laboratory. It was prepared by Dr. D. Opresko of the Oak Ridge National Laboratory and benefited from technical review by members of the Working Group and by Dr. J. Moya of the National Center for Environmental Assessment (EPA).

The ERAP Advisory and Coordinating Committee endorses the information contained within this document with the understanding that the end user is responsible for its application. This means that users are responsible for obtaining any internal scientific and policy reviews required prior to its acceptance within other organizations.

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1. INTRODUCTION

EPA guidelines for exposure assessments appeared in the Federal Register in May 1992 (57FR:22888-22938). These guidelines, which supersede those issued in 1986 (U.S. EPA, 1986a), review the general concept of exposure assessments and include a standardization of definitions and associated units, descriptions of how exposure assessments should be planned and conducted, and how the results should be presented.

As noted in the guidelines, exposure assessments can be used in various ways, such as in studies evaluating status and trends of exposure (e.g., occupational exposure situations); in epidemiological studies to establish dose-incidence or dose-effect relationships; and as one of the four major components of risk assessments (the others being hazard identification, dose-response assessment and risk characterization [NRC, 1983]). This report will focus on the use of exposure assessments as an integral part of risk assessments.

The type of exposure assessment that is undertaken is dependent on the primary objective of the risk assessment. In risk assessments that are used to determine whether waste sites should be remediated, the emphasis is placed on calculating the health risk to a population by comparing an estimate of the potential exposure (or dose) with a reference toxicity value which, for noncarcinogens, is defined as "an estimate (with uncertainty spanning perhaps an order of magnitude or greater) of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime" (U.S. EPA, 1989a). For carcinogens, the reference toxicity value is the slope factor which is the "plausible upper-bound estimate of the probability of a response per unit intake of a chemical over a lifetime" (U.S. EPA, 1989a). If there is the possibility that exposures are currently occurring, the guidelines indicate that the use of personal monitoring (or measurements of biomarkers, metabolites or body burdens from which exposures might be back calculated) can be used to estimate exposure. However, if the primary concern is the potential health risks associated with future use of a site after remediation or in the absence of remediation, then potential exposure must be estimated using one or more exposure scenarios. In such cases the primary objective of the exposure assessment is linking potential sources of contamination with individuals and populations who might be exposed in the future. Therefore, a key issue in setting cleanup standards (i.e., control limits or acceptable residual levels) at a particular site is in the identification of the most likely future use of the site. This will identify the populations that are most likely to be at risk and the length of time that they may be at risk, both of which will affect the population-specific parameters used in estimating exposure. Population-specific parameters are used in conjunction with site- and chemical-specific parameters to derive estimates of potential exposure. When information in any of these areas is lacking, default values based

on professional judgement, are substituted. These default values can represent averages, ranges, or distributions of values of the parameter. The choice of default values affects the level of conservatism and the level of uncertainty associated with the final adopted cleanup standards.

This report reviews the EPA exposure assessment guidelines issued in the 1992 Federal Register notice (U.S. EPA, 1992a) as well as those described in the Superfund Manual (RAGS) (U.S. EPA, 1989a). Emphasis is placed on describing the general concepts used in exposure assessments and in identifying the population-, site- and chemical-specific exposure parameters that are recommended in these sources. In addition, alternative approaches recommended by other governmental agencies and by non-governmental research groups are also included. The report concludes with a summary of the key issues in exposure assessment methodology.

The primary objective of this report is to identify for SERDP those steps in the current exposure assessment process for which there might be alternative approaches more appropriate for SERDP needs, and which may be may lead to an overall improvement in the cost efficiency and quality of the exposure assessments conducted on DOD or DOE sites.

It should be noted that methodologies for evaluating indirect exposures, such as those associated with point source atmospheric pollution from waste combustion facilities, are not addressed in this report. The reader is referred to U.S. EPA (1990c) for specific information on indirect exposure assessments.

2. GENERAL APPROACH

An exposure assessment consists of five major components (U.S. EPA, 1986):

- Source Assessment. The identification of the potential sources of exposure (i.e., the environmental media that may be contaminated as a result of land disposal, release of aqueous effluents, air emissions from production or incineration stacks, and open burning, detonation or testing of military items).

- Pathways and Fate Analysis. The description of how the contaminant may be transported from the source to the potentially exposed population under current conditions and at various time points in the future. The identification of the major routes of exposure (i.e., ingestion, inhalation, dermal contact).

- Contaminant Concentration Analysis. The use of monitoring data and/or modeling methods to estimate contaminant concentrations at on-site or off-site locations where populations may be exposed.

- Population Analysis. A description of the size, location, and habits of potentially exposed populations as identified through the most likely exposure scenarios and exposure pathways. Identification of potential impacts to the most sensitive subpopulations such as children, aged, hospital patients, etc.

- Integrated Exposure/Dose Analysis. The calculation of exposure/dose levels for current, past, or future use scenarios for potentially exposed populations onsite or offsite (through transport of contaminants across site boundaries). Identification of the most appropriate value/s for characterizing exposure (e.g., average, median, 90th percentile). Qualitative and quantitative uncertainty analysis of the exposure estimates.

Primary sources of exposure are identified through examination of records of past disposal practices, monitoring of current conditions, or modeling future conditions. Fate and transport models are used to predict the possibility of secondary sources of contamination resulting from the movement of the chemicals through environmental media, and analytical and numeric models are used to estimate concentrations of contaminants in bulk media at various distances from the sources of the contamination. Population analysis involves: (1) characterizing past, current and/or future uses of the site; (2) identifying exposure pathways relevant to onsite or offsite populations that may have been, are currently, or who might be exposed in the future to contaminants originating on the site; (3) identifying the physical and behavioral characteristics of the potentially exposed population, and (4) estimating the expected frequency and duration of the exposures.

Information on the contaminant concentrations and the characterization of the exposed populations are combined to estimate the average daily exposure for each relevant pathway. Exposures for a specific pathway or for several combined pathways are compared with the appropriate reference toxicity values to determine whether any segment of the exposed population is at risk.

2.1 SOURCE ASSESSMENT

If an exposure assessment is focused on past exposures, a source assessment should begin with an examination of invoices and records and a mass balance analysis to determine how much material was brought onsite, how much was consumed or disposed of, and how much might still be present. In the absence of information on previous levels of release, and for evaluations of current or future exposures, the source assessment involves the

characterization of the physical setting of the site and the identification of the contaminants, and/or their degradation or transformation products in various environmental media. Important site characteristics include: climate (temperature and precipitation); meteorology (wind speed and direction); geologic features; vegetation; soil type; ground-water hydrology, and surface water characteristics (U.S. EPA, 1989a). Contaminants at the site can be identified using analytical methods and/or modeling. Monitoring studies can be conducted on a variety of environmental media depending on the likely sources of contamination and the objectives of the exposure assessment (U.S. EPA, 1989b; 1992a). These include:

- fixed location monitoring of air or water quality
- short-term media monitoring
- source monitoring of facilities
- food sampling
- drinking water sampling
- consumer products
- breathing zone measurements
- microenvironmental studies
- surface soil sampling
- soil cores
- fish tissue sampling

Important steps in establishing monitoring/sampling programs for exposure assessments include the following (U.S. EPA, 1989b; U.S. EPA, 1992a):

- setting Data Quality Objectives (U.S. EPA, 1987a,b; 1993d)
- developing a sampling plan and design
- comparing samples against spiked and blank samples,
- assessing background levels
- validating previously generated data
- conducting a Quality Assurance/Quality Control program
- selecting the most suitable method of analysis
- identifying uncertainties associated with sampling, analysis, and data manipulation

Data quality objectives (DQO) describe the acceptable degree of uncertainty associated with the inherent variability of the quantities being measured (spatial and temporal) as well as the variability of the analytical measurements. DQOs should consider data needs, cost-effectiveness, and the capability of the measurement process. Superfund guidance on DQOs (U.S. EPA, 1987a,b) has recently been revised (U.S. EPA, 1993b) to be consistent with other Agency-wide Quality Assurance requirements and guidance documents (U.S. EPA, 1992e; 1993d,e).

The sampling and analysis plan specifies how a sample is to be selected and handled. Sampling design specifies the number, location and types of samples needed to achieve

DQOs. A statistical or environmental process model may be used to allocate sampling effort in the most efficient manner (U.S. EPA, 1992a). General EPA guidelines for data collection are available (U.S. EPA, 1984b; 1993f). Specific guidance is also available for soil (U.S. EPA, 1989f; 1990a), groundwater (U.S. EPA, 1985d), and stream sampling (U.S. EPA, 1986b). Guidance for characterizing populations is contained in "Methods for Assessing Exposure to Chemical Substances" (U.S. EPA, 1985e). In selecting sampling locations, the factors that should be considered include: population density, historical sampling results, patterns of environmental contamination, and environmental characteristics such as stream flow, wind direction, access to the site, types of samples and health and safety requirements (U.S. EPA 1992a). Statistical methods for selecting sampling locations are described by Schweitzer and Black (1985) and Schweitzer and Santolucito (1984). Sampling frequency is determined by the physical and chemical characteristics of the contaminant, the type of health effect being evaluated (acute or chronic), the ways in which the contaminant is being released into the environment, how rapidly contaminant patterns are changing, and to what degree physical conditions are expected to vary in the future (U.S. EPA, 1992a). Budget constraints may also influence the total number of samples taken. Statistical techniques for determining the optimal number of samples are discussed by Shaw et al. (1984), Sanders and Adrian (1978) and Nelson and Ward (1981). Sampling duration depends on the analytical method chosen, the limits of detection, the physical and chemical properties of the analyte, chemical concentration, and knowledge of transport and transformation mechanisms (U.S. EPA, 1992a). Sampling duration is directly related to selection of statistical procedures, such as trend or cross-sectional analysis (U.S. EPA, 1992a). Samples should be analyzed as soon as possible after collection to avoid storage stability problems, unless storage stability has been previously documented.

Sample validity should be confirmed by the concurrent analysis of field- and laboratory-spiked and blank samples. Any data not significantly greater than blank samples should be used with considerable caution (U.S. EPA, 1992a). The following guidance is offered by EPA:

- For volatiles and semivolatiles, the amount should exceed 5 times the amount in the blank, with the exception of the common laboratory contaminants methylene chloride, acetone, toluene, 2-butanone, and common phthalate esters which should exceed 10 times the amount in any blank.

- For pesticides and PCBs, the amount should exceed 5 times the amount in the blank.

- For inorganics, the amount should exceed 5 times the amount in the blank.

Background levels of contaminants may be present as a result of natural or anthropogenic sources (U.S. EPA, 1992a). Background concentrations should be determined, using the same level of analytical detail, on samples taken from nearby locations unaffected by the site.

A Quality Assurance program begins with the DQOs and includes the development and strict adherence to principles of good laboratory practice, consistent use of standard operational procedures, and carefully designed protocols for each measurement effort (U.S. EPA, 1992a). The program should insure that errors be statistically characterized and reduced to acceptable levels. EPA guidance for the preparation of a Quality Assurance program is available (U.S. EPA, 1980; 1993e,g). Previously collected data used in current assessments should be subjected to Quality Assurance validation, including an evaluation of collection protocol, analytical methods, detection limits, laboratory performance, and sample handling. A determination should be made as to whether the data are still representative of the site.

Analytical methods are selected based on their appropriateness for the contaminants of concern. New analytical methods require field and laboratory testing and validation. EPA guidance on the selection of analytical methods is available (U.S. EPA, 1990b).

After the data have been collected and validated, they must be evaluated to determine whether the DQOs have been satisfied by conducting a Data Quality Assessment (U.S. EPA 1993g). A Data Quality Assessment involves the use of statistical tools to determine whether the variability and bias in the data are small enough to allow the site manager to use the data to support the decision with acceptable confidence.

In the absence of monitoring data, possible sources of exposure can be determined from simple conceptual models based on the available site sampling data, historical records, aerial photographs, and site hydrogeologic information (U.S. EPA, 1988a, 1989a). Basic physical and chemical information incorporated into models can be used to prioritize chemicals, focus exposure assessments on specific environmental media, and, when used in conjunction with generic assumptions about exposure pathways, establish chemical-specific screening levels (U.S. EPA, 1994d, 1994g, 1994h).

A modeling strategy has several aspects, including the setting of objectives, model selection, obtaining and installing the code, calibrating and running the model, and validation and verification (U.S. EPA, 1992a). The objectives should focus on how the model can help address the questions or hypotheses of the exposure assessment. If the model is used to simulate pollutant behavior at a site, data specific to the site, such as source characterization, dimensions and topography of the site, location of the receptor populations, meteorology,

soils, geohydrology and ranges and distribution of the contaminants, should be incorporated into the model whenever possible. In the absence of site data, reasonable default values should be used (U.S. EPA, 1992a). For exposure models that simulate both chemical concentration and time of exposure (through behavior patterns), data on both of these parameters must be evaluated (U.S. EPA, 1992a). If necessary, the source code for the model must be adapted for the computer on which it is run, without affecting the way the model runs or the results it produces. Calibration of the model is the process of adjusting selected model parameters within an expected range until the differences between model prediction and field observations are within selected criteria. Model validation is the process by which the accuracy of the model is compared with actual data from the system being simulated. Uncertainties in the model output, which may result from deficiencies in either the model itself or in the input data used in the model (i.e., estimated properties or rates) should be identified and characterized.

2.2 PATHWAYS AND FATE ANALYSIS

After a chemical is released to the environment, it may be: transported through the medium; physically transformed (e.g., volatilization, precipitation); chemically transformed (e.g., hydrolysis, oxidation); biologically transformed (e.g., biodegraded); or accumulated in one or more media (U.S. EPA, 1989a). For contaminants in soils, transport between media can occur through various mechanisms including volatilization, fugitive dust generation, surface runoff, leaching to ground water, or direct uptake by biota (U.S. EPA, 1989a).

Evaluations of the environmental fate of contaminants can be made from (1) field measurements (i.e., field degradation studies); (2) laboratory studies (i.e., soil column studies, biodegradation tests, ecosystem microcosms); or (3) physical and chemical data (water solubility, octanol/water partitioning coefficient, Henry's Law Constant, etc.) to estimate environmental fate parameters (i.e., soil adsorption, bioconcentration) using simple regression equations (Lyman et al., 1982) and/or environmental transport and fate models. Such information, can be used to focus an exposure assessment on certain exposure pathways and/or exclude others from consideration. The following are selected examples of the use of such information:

- Laboratory soil column studies can indicate the potential for contamination of groundwater; however, unless the site conditions (i.e., type of soil, organic content, pH, temperature) are duplicated, some level of extrapolation is required to apply the results to a specific location. Site-specific information on climatology, geology and hydrology would also be needed to determine how local conditions such as rainfall, or geological features might affect movement of the contaminants through the soil.

- Octanol/water partitioning coefficients (K_{ow}) can be used to estimate the soil-organic carbon partitioning coefficient (K_{oc}). K_{oc} is an estimate of the partitioning of the substance between the organic carbon and water phases in the soil. Adsorption of contaminants to non-organic minerals may be more important than K_{oc} under certain conditions; i.e., (1) in clays with very high surface area, (2) where cation exchange (e.g., for dissociated organic bases) occurs, (3) where clay-colloid-induced polymerization occurs, and (4) where chemisorption is a factor (Lyman et al., 1982). Methods for estimating soil adsorption under such conditions are not currently available.

- Environmental partitioning models based on chemical/physical parameters such as K_{oc} and Henry's Law Constant are available to estimate the distribution of a contaminant between various soil compartments. Such models normally use default values for site parameters such as soil porosity, temperature, pH, and organic carbon content. Models incorporating site-specific data would provide more valid results, but these may still not include all relevant partitioning processes. A description of one type of soil equilibrium partitioning model using K_{oc} and Henry's law Constant is provided in Appendix A.

- Soil Sorption Coefficient (K_d) is a chemical and soil specific value which is used for determining the chemical's mobility, fate and transport through soil. K_d is an estimate of the partitioning of a substance between soil and water.

- Water solubility data can indicate the potential for surface or ground water contamination; however, regional climatic conditions may preclude such transport.

- Vapor pressure can indicate the potential for atmospheric contamination; however, in some cases, atmospheric contamination from wind-blown dusts may be more important than volatilization.

- Henry's Law constant can indicate the extent of partitioning between air and water at equilibrium.

- Plant uptake models can be used to predict the contaminant uptake from soil to plants. Some models estimate plant uptake based on K_{ow} (Calamari et al., 1987). EPA recommends that the methodology of Briggs et al. (1982) be used to estimate plant uptake of non-ionized organic contaminants. For heavy metals, EPA suggests using the data presented by Sauerbach (1988) for a qualitative assessment of plant uptake.

- Bioconcentration in aquatic organisms can be predicted from octanol/water partitioning coefficients (K_{ow}) for organic compounds (U.S. EPA, 1989a).

- Bioconcentration in terrestrial organisms can be estimated from transfer coefficients (Ng et al., 1982; Baes et al., 1984). EPA (1991b) recommends that the procedures described in Travis and Arms (1988) be used to estimate concentrations of organic contaminants in beef and milk.

A summary of pertinent environmental fate parameters and the possible interpretations of such data for screening purposes is given in Appendix B.

2.3 CONTAMINANT CONCENTRATION ANALYSIS

In an exposure scenario evaluation, the estimation of pollutant concentrations at the expected points of contact with the exposed population is usually carried out by using a combination of monitoring data and mathematical modeling (U.S. EPA, 1989a).

2.3.1 Monitoring Data

For assessments of current conditions, field data may include fixed location monitoring and/or short-term media monitoring involving the analysis of soil, drinking water, surface water, groundwater, indoor and outdoor air, locally grown produce, and locally raised or hunted food animals.

The Superfund Guidelines state that the contaminant concentration in an environmental medium that is to be used to estimate exposure should be the 95th percent upper confidence limit (UCL) on the arithmetic mean concentration, but that this value should not be used if it is higher than the maximum detected concentration. A subsequent EPA/OSWER directive has reiterated the importance of using the 95% UCL because of the uncertainty associated with estimating the average concentration at a given site (U.S. EPA, 1992f).

Data sets for contaminant concentrations may contain values below analytical detection limits (referred to as censored data). Because these values, are not necessarily equal to zero, they are often estimated by the use of a simple substitution, a distributional method, or a robust method (U.S. EPA, 1992a). Substitution methods involve the use of a single value (i.e., the detection limit or one-half the detection limit) as a proxy for each nondetected data value. Distributional methods identify the distribution type (e.g., lognormal) for all data points above the detection limit and assume that all the points below the detection limit follow the same distribution. Robust methods generally assume a distribution form for the nondetected data points rather than for the entire data set, and extrapolation from the nondetected data points is done by regression analysis. When censored data is used in an assessment, EPA requires that these values be clearly labeled as hypothetical, and that any exposure, dose or risk estimations be reported as "less than" the calculated value. RCRA guidelines state that when a criterion is below detection limits, the detection limit should be used, unless values can be derived from acceptable models (U.S. EPA, 1989e).

The Superfund Guidelines discuss several general issues relating to the monitoring of contaminant concentrations in ground water, soil, air, surface water, sediments and food; several of these are summarized below:

Groundwater - Estimates of future concentration of contaminants in ground water that are based on current monitoring studies should assume steady-state conditions (U.S. EPA, 1989a). EPA Region IV states that data points for calculating groundwater exposure point concentrations for a future-use scenario should consist of wells located within the contamination plume (U.S. EPA, 1991e).

Soil - Soil monitoring data and site conditions should be carefully screened to determine whether contaminant concentrations may decrease over time. If models are not used to estimate this decrease, the contaminant concentration should be assumed to be constant over time (U.S. EPA, 1989a). Exposure point concentrations for soils should be determined based on the spatial contaminant distribution. If the contaminant is widely dispersed throughout the site, the exposure point concentration should be based on the 95% UCL of the arithmetic average for all site samples including non-detects. However, if the contamination is unevenly distributed, i.e., "hotspot" areas exist, these areas should be evaluated separately. A percentage of the time on the site that would be spent in these areas should be factored into the exposure estimate (U.S. EPA, 1991e). Surface and subsurface soil samples should be evaluated separately if direct contact with surface soil or inhalation of wind-blown dust are potential exposure pathways (U.S. EPA, 1989a).

Air - Air monitoring data can be used to derive exposure concentrations if there are no significant analytical problems affecting the sampling data, if background levels are not significantly higher than site-related levels, and if site-related levels are not below detection limits (U.S. EPA, 1989a).

Surface Water - Estimates of exposure based on monitoring of surface waters should take into account temporal changes brought about by seasonal or episodic (i.e., storms) alterations; spatial differences caused by size and amount of mixing; dilution to below detection levels; and contributions from other sources.

Sediments - Sediments may provide better temporal representativeness of potential exposure than surface water concentrations. Direct contact scenarios should consider surficial, near-shore sediments.

Food - Site-specific measured values are preferable to estimated values and should characterize the population and the species of concern.

2.3.2 Models

In the absence of monitoring data, or in developing future-use exposure scenarios, models for environmental fate and transport may be used to estimate exposures based on predicted residual concentrations of contaminants in various media.

Models may be required to estimate exposures when:

- known or potential exposure points are spatially separated from monitoring points.
- changes in concentrations may occur over the long-term.
- contaminant concentrations occur below the limit of quantification.

According to the Exposure Assessment Guidelines, model selection should be determined by the objectives of the study, the technical capabilities of the model, how readily the model can be obtained, and how difficult it is to use (U.S. EPA, 1992a). If models are used to simulate pollutant behavior at a specific site, all available information on source characterization, dimensions and topography of the site, location of the receptor populations, meteorology, soils, geohydrology and ranges and distributions of chemical concentrations must be considered (U.S. EPA, 1992a).

All models have specific ranges of application and specific classes of chemicals for which they are appropriate. The model must adequately represent the physical setting of the site and simulate the relevant transport and transformation processes occurring at the site. Field data from the area where the exposures are to be estimated should be used to define the input parameters. In cases in which these data are not available, parameter values representative of field conditions should be used as defaults. Assumptions of homogeneity and simplification of site geometry may allow use of simpler models. EPA notes that the use of estimated properties or rates in models adds to the uncertainty in the exposure concentration estimates, and that these uncertainties and the validation status of the model must be clearly indicated (U.S. EPA, 1992a).

A wide variety of models are available for use in exposure assessments. Simple models may consider only mass balance, dilution, dispersion, or equilibrium partitioning (see Appendix C). More complex models may analyze environmental fate processes such as chemical reactions, biodegradation, and photolysis. General descriptions of models and guidance in selecting models is provided in the Superfund Exposure Assessment Manual (U.S. EPA, 1988c) and in the Exposure Assessment Methods Handbook (U.S. EPA, 1989d). EPA has also developed the Integrated Model Evaluation System (IMES) for evaluating and selecting environmental fate and transport models. Detailed evaluations of surface water, ground water and soil models are also available (U.S. EPA, 1987c, 1988b, 1991a, 1993i, 1994e, 1994f).

2.4 POPULATION ANALYSIS

The primary objective of exposure assessments evaluating human health effects is the linking of potential sources of contamination with individuals and populations who might be exposed. In the exposure assessment guidelines described in the Superfund Manual (U.S. EPA, 1989a), the first step in this process involves the characterization of the exposure setting with regard to the current and future populations who might be exposed. Information on current land use patterns and/or zoning restrictions in the immediate vicinity of a site can provide an indication of how the site might be used if it is made available to local governments or private developers.

2.4.1 Exposure Scenarios

Possible exposure scenarios range from the occasional trespasser at a restricted site that has not been remediated to permanent residential occupancy of a former site at some time point following remediation. Some of the possible exposure scenarios that might be considered in an exposure assessment are listed below:

1. Residential Use.
    a. Full-time single or multifamily homes with lawn and garden areas.
    b. Seasonal single or multiple family homes with lawn and garden areas.
    c. Apartments, condominiums, hotel/motels with limited lawn or garden areas.
    d. Family farms
2. Educational Use.
    a. Preschool or day care programs or grade schools with outdoor playgrounds.
    b. Higher grade schools with athletic fields.
3. Agricultural Use.
    a. Food crops
    b. Non-food crops
4. Livestock operations.
    a. Food animals.
    b. Non-food animals.

5. Industrial Use.

   a. Facilities located at or nearby the site
   b. Construction or demolition activities at the site.

6. Recreational Use.

   a. Swimming, boating
   b. Hunting, camping, fishing.

7. Restricted Use.

   a. No public or governmental access allowed (for limited or indefinite time period).
   b. Governmental access only, for scientific or military use.
   c. Maintenance activities at the site

EPA Approach - The Superfund Guidelines (U.S. EPA, 1989a) state that population locations and activity patterns in the area should be considered in identifying possible exposure scenarios, particularly for subgroups, such as children, that might be of special concern. Any contaminated media or sources onsite can be a potential exposure point if the site is currently being used, if access to the site under current conditions is not restricted or otherwise limited, or if contact is possible under an alternate future land use (U.S. EPA, 1989a). The Superfund Guidelines specifically address the methodology for quantifying exposures for residential settings; however, general guidance is also given for deriving exposure estimates for occupational/industrial and recreational settings. EPA Region IV guidance is that "a future residential scenario should be assumed unless there is strong reason to do otherwise, i.e., highly industrial area or wetlands" (U.S. EPA, 1991e).

2.4.2 Exposure Pathways

Within each exposure scenario there can be one or more relevant exposure pathways. Not all pathways will be applicable to each exposure scenario, and of those that apply to a specific scenario, not all will be equally important. In some cases, information on the environmental fate of the contaminants and/or on the activity patterns of the population can be used to eliminate some pathways from consideration; however, preliminary estimates of potential exposure may be required to evaluate the importance of other pathways.

EPA Approach - In Part A of the Superfund Guidelines for baseline risk assessments (U.S. EPA, 1989a), the following residential exposure pathways are evaluated:

- Ingestion of chemicals in drinking water and beverages made using drinking water.
- Ingestion of chemicals in surface water while swimming.
- Dermal contact with chemicals in water.

- Ingestion of chemicals in soil.
- Dermal contact with chemicals in soil.
- Inhalation of airborne (vapor-phase) chemicals during showering.
- Ingestion of contaminated fish and shellfish.
- Ingestion of contaminated fruits and vegetables.
- Ingestion of contaminated meat, eggs and dairy products.

In addition to these pathways, the Exposure Factors Handbook (U.S. EPA, 1990d) also addresses the issue of inhalation of airborne vapors and particulate matter inside and outside the home. Part B of the Superfund Guidelines deals with the derivation of preliminary remediation goals (PRG) using the following exposure pathways (U.S. EPA, 1991c):

- Ingestion of chemicals in drinking water derived from surface or ground water at residential sites.
- Ingestion of chemicals in soil at residential and commercial/industrial sites.
- Inhalation of volatiles released from soil at commercial/industrial sites.
- Inhalation of particulates released from soil at commercial/industrial sites.

Three of the exposure pathways used in the PRG process have also been used by EPA to develop soil screening levels for specific pollutants (U.S. EPA, 1994d, see also Section 2.5.6). EPA Region III has adopted screening levels based on contaminant intake from inhalation and from ingestion of drinking water, fish, and soil (residential and commercial/industrial sites) (U.S. EPA, 1993c).

2.4.3 Exposure Parameters

Population-specific parameters that need to considered in evaluating specific exposure pathways include: physical factors (e.g., body weight, body surface area); activity patterns which define exposure frequency and duration (e.g., time of residence, time spent outdoors, time spent gardening, etc); and intake factors (rates of inhalation and ingestion of drinking water, foods, and soil).

In the absence of site-specific data on a given exposure parameter, EPA uses default values which can represent averages, ranges, or point estimates derived from distribution data. Where distributions of values are available, the 50th percentile value is used as the central tendency (CT) and values above the 90th percentile are used as the reasonable maximum exposure (RME) (see Section 2.5.2). The Superfund Guidelines recommend using the 90th or 95th percentile for the default contact rate, exposure frequency, and exposure duration variables (U.S. EPA, 1989a). The Exposure Assessment Guidelines specifically refer to EPA's Exposure Factors Handbook (U.S. EPA, 1990d) as the primary source for the default values to use for intake and uptake parameters; however, for Superfund risk assessments, two directives have appeared since 1990 containing revised default values for some

parameters (U.S. EPA, 1991b, 1993a). In addition, EPA Regional Offices have the authority to specify default values different from those listed in the Superfund Guidelines for use within their region. A brief summary of the recommended EPA default values and those of other government agencies and non-governmental groups is given in the following sections.

2.4.3.1 Body Weight

Body weight is used in exposure calculations to normalize dose to unit weight.

EPA Default Values - The default value for average adult body weight (males and females combined) that is used in Superfund risk assessments and in Drinking Water Health Advisories is 70 kg (U.S. EPA, 1993a, 1994a). Body weight distributions are given in the Exposure Factors Handbook (U.S. EPA, 1990d): the 50th percentile weights for men and women are 75.9 kg and 61.5 kg, respectively; the corresponding 90th percentile weights are 94.7 and 83.4 kg, and 95th percentile weights are 101.7 and 92.3 kg.

In Superfund risk assessments, an average body weight of 16 kg has been used as a default value for children 1-6 years old, a group considered to be most susceptible to ingestion of contaminated soil (U.S. EPA, 1989a). This value corresponds to the 50th percentile group. A value of 15 kg is recommended for children under 6 years of age (U.S. EPA, 1993a). In deriving Drinking Water Health Advisories, EPA uses a standard body weight of 10 kg for children (U.S. EPA, 1994a). Fiftieth-percentile, age-specific body weights for boys and girls, as tabulated by EPA (1985a) from the results of the second National Health and Nutrition Survey, are shown in Table 2-1.

Other Values - Standard reference body weights used by other federal and state agencies are listed in Table 2-2, together with the EPA default values. The International Commission on Radiological Protection (ICRP) "Report of the Task Group on Reference Man" (ICRP, 1975) reported that the average body weight of adult females is 58 kg and that of adult males is 70 kg. ICRP (1975) also reported an average body weight of 10 kg for children 1-3 years old and 27 kg for children 3-10 years old.

Brainard and Burmaster (1992) examined height and weight data obtained by the U.S. Public Health Service during the second National Health and Nutrition Examination Survey (NHANES II) (National Center for Health Statistics, 1987) and fit bivariate distributions to the values for men and women. From the bivariate analysis, it was estimated that the arithmetic average weight was 78.6 kg for men and 65.8 kg for women. Brainard and Burmaster (1992) suggested that the distributions of height and weight would be suitable for use in risk assessments using Monte Carlo simulations.

Table 2-1. Age-specific Body Weights (kg) for Children - 50th Percentile[a]			
Age	Boys	Girls	Both Sexes[b]
<1	9.2 (0.16)	8.5 (0.09)	8.85
1 < 2	11.5 (0.12)	10.5 (0.08)	11.0
2 < 3	13.4 (0.07)	12.6 (0.03)	13.0
3 < 4	15.3 (0.10)	14.6 (0.15)	14.95
4 < 5	17.4 (0.10)	16.4 (0.27)	16.9
5 < 6	19.3 (0.12)	18.8 (0.17)	19.05
6 < 7	21.9 (0.10)	21.0 (0.83)	21.45
7 < 8	24.4 (0.30)	23.5 (0.43)	23.95
8 < 9	27.3 (0.69)	27.3 (0.81)	27.3
9 < 10	29.7 (0.88)	29.6 (0.45)	29.65

Source: U.S. EPA, 1985a (data points are also available for children 10-18 years old).
[a] Standard error shown in parentheses.
[b] Calculated in this report as the average of the median values for boys and girls.

Table 2-2. Standard Reference Body Weights (kg) Used by State and Federal Agencies					
Age Group	EPA Superfund	ATSDR[a]	U.S. DHHS[b]	Mass.[c]	Calif.[d]
Infants		10	3.4		6.5
0-2					10
1-6	16	16			
2-10					20
11-18					40
All children				10	
Adults	70	70		70	

[a] ATSDR, 1990
[b] U.S. DHHS, 1970
[c] MDEQE, 1989
[d] CDHS, 1986; CAPCOA, 1987; CAPCOA, 1990

Finley et al. (1994) used data from the NHANES II study (National Center for Health Statistics, 1987) to derive a cumulative density function for age-specific body weights for four age classes. They estimated the 50th and 95th percentiles (for adult men and women combined) to be 70 kg and 101 kg, respectively. Finley et al. (1994) also developed body weight distributions for children in 1-year increments. Arithmetic mean weights were as follows: 9.4 kg for children 0.5-1 year old; 11.8 kg for children 1-2 years old; 13.6 kg for children 2-3 years old; 15.7 kg for children 3-4 years old; 17.8 kg for children 4-5 years old; and 20.1 kg for children 5-6 years old.

2.4.3.2 Body Surface Area

In dermal exposure situations, the level of exposure is dependent on the amount of skin surface area available for contact which is a function of the age, body size, and activity pattern of the individual.

EPA Default Values - The Superfund Guidelines recommend using 50th percentile body surface area values (as complied by U.S. EPA, 1985a and U.S. EPA, 1990d) for estimating dermal exposures. These values are shown in Table 2-3. The 95th percentile values for total body surface area are 20,900 cm^2 for adult women and 22,800 cm^2 for adult men (U.S. EPA, 1990d). Age-specific data for surface areas of the arms, hands and legs for male children 3-10 years old are also shown in Table 2-3. Additional information on body surface areas is provided in the EPA document, Dermal Exposure Assessment: Principles and Applications (U.S. EPA, 1992b).

Table 2-3. Body Surface Areas (m^2) - 50th Percentile Values

Total body			Body parts (males only)			
Age (yr)	Male	Female	Age (yr)	Arms	Hands	Legs
3 < 6	0.728	0.711	3 < 4	0.096	0.040	0.18
6 < 9	0.931	0.919	6 < 7	0.11	0.041	0.24
9 < 12	1.16	1.16	9 < 10	0.13	0.057	0.31
12 < 15	1.49	1.48	-	-	-	-
12 < 18	1.75	1.75	-	-	-	-
Adults	1.94	1.69	Adults	0.23	0.082	0.55

Source: U.S. EPA, 1989a, 1990d, 1985a.

Other values - Body surface areas of 19,400 cm^2 for adult males and 16,900 cm^2 for adult women are also used as standard reference values by ATSDR and a number of state agencies (ATSDR, 1990, WDOE, 1990; MDEQE, 1989). ATSDR recommends using the following values for children: 3500 cm^2 for children 0-1 years old; 8750 cm^2 for children 1-11 years old; and 15,235 cm^2 for children 12-17 years old.

Reference body surface area values recommended by ICRP (1975) include the following: 2,115 cm^2 for newborn infants; 3,925 cm^2 for older infants; 16,000 cm^2 for adult women; and 18,000 cm^2 for adult men.

Phillips et al. (1993) determined that the ratios of total skin surface area to body weight are lognormal distributed for children 0-2 years old and normally distributed for older children and adults. The reported arithmetic means (and standard deviations) were 641 cm^2/kg (SD = 114 cm^2/kg) for children 0-2 yr old; 423 cm^2/kg (SD = 76 cm^2/kg) for children 2-18 yr old; and 284 cm^2/kg (SD = 28 cm^2/kg) for adults.

Based on data tabulated in the Exposure Factors Handbook (U.S. EPA, 1990d), Finley et al. (1994) developed age-specific distribution factors (maximum, minimum and mean) for the fraction of total skin surface area associated with body parts important in dermal exposures evaluations (head, arms, hands, legs, and feet).

2.4.3.3 Exposure Frequency and Duration

Exposure frequency and/or duration will vary with the type of exposure scenario under consideration, as well as with temporal and spatial changes in the activity patterns of the individuals exposed. Consequently, estimates of exposure frequency/duration can be the single largest source of uncertainty in an exposure assessment (U.S. EPA, 1992a). Information on likely patterns of exposure can be obtained by the use of census and demographic data, survey statistics, behavioral observations, activity diaries, activity models, or, in the absence of more substantive information, by the use of general assumptions about behavior. For exposure scenarios relating to residential situations, key factors that must be considered are: life expectancy, time of residency at one location, and time spent indoors and outdoors at that location. Regional variations in time use patterns can affect estimates of exposure duration and may be important in exposure assessments. For example, populations in the South and West regions of the U.S. are likely to spend more time outdoors than population in the North Central or Northeast regions. EPA states that estimates of exposure duration "based on data collected close to the actual point of contact are preferable to those based on indirect measurements; both of these are preferred to estimates based on assumptions alone" (U.S. EPA, 1992a).

EPA Default Values:

Life Expectancy. In carcinogenic health assessments in which lifetime exposures are evaluated, EPA uses 70 years as the current default value for life expectancy. However, 1985 census data indicate that life expectancy for the total U.S. population is 74.7 years, 71.2 for males and 78.2 for females (U.S. EPA, 1990d). Although the Exposure Factors Handbook considers 75 years to be a more appropriate average value for life expectancy in the U.S., 70 years is still used for Superfund risk assessments in order to compare values with cancer slope factors based on the 70-yr value (U.S. EPA, 1991b).

Residential Occupancy. Using information obtained in a survey conducted in 1983 by the Bureau of the Census, EPA calculated that the median length of time that a homeowner lives in the same house (50th percentile of the distribution) is about 9 years (U.S. EPA, 1990d). The 90th percentile of the distribution is about 30 years. Nine and 30 years are the central tendency (CT) and reasonable maximum exposure (RME) values recommended for use in Superfund risk assessments (U.S. EPA, 1993a). For Superfund risk assessments, EPA proposes default values of 234 days/year (CT) and 350 days/year (RME) for the time spent at home (U.S. EPA, 1993a).

Occupational Exposures. For occupational exposures, a default value of 25 years is proposed by EPA as a reasonable maximum exposure. This value is based upon the 95th percentile for the number of years worked at the same location, as reported by the U.S. Bureau of Labor Statistics (1990). A exposure frequency of 250 days/year has been proposed as the default RME (U.S. EPA, 1993a).

Activity Patterns. The Exposure Factors Handbook cites the results of several national surveys (Szalai, 1972; Robinson 1977; Juster et al., 1983; Juster, 1985) as sources of information on the expected duration of exposure for basic time-use activities, such as time spent indoors at home. Selected data from these survey, as reported by EPA (U.S. EPA, 1990d) are shown in Tables 2-4 and 2-5. However, EPA notes that these data sets "may or may not be appropriate for any given assessment" (U.S. EPA, 1990d).

Time Spent Showering. For estimating exposures resulting from inhalation of volatile compounds released from water during showering, the Exposure Factors Handbook recommends using the data of James and Knuiman (1987) which indicate that the median shower length is approximately 7 minutes and the 90th percentile is approximately 12 minutes (U.S. EPA, 1990d). In evaluating the available data (including that of Jo et al., 1990), EPA concluded that "If time, resources, or other constraints prevent the use of modeling or monitoring data as recommended, risk assessors should assume that exposure to VOCs during showering is equivalent to exposure from ingesting two liters of the same water per day" (U.S. EPA, 1991d).

Table 2-4. Time-Use Patterns (weighted mean hr/wk)			
Activity	**Men**	**Women**	**Men & Women**
Time spent outdoors	5.28	2.77	3.91
Time spent indoors	152.87	157.11	155.17
Time spent at home	97.80	115.98	107.59
Time spent outdoors at home	4.17	2.13	3.07
Time spent away from home	70.27	52.10	60.49
Gardening/pet care	0.94	0.72	0.97

Source: Adapted from U.S. EPA, 1990d

Table 2-5. Activity Patterns (average hr/day)		
Location	**Activity**	**Hr/day**
Indoors:	Resting	9.82
	Light	9.82
	Moderate	0.71
	Heavy	0.098
	Total	20.4
Outdoors:	Resting	0.505
	Light	0.505
	Moderate	0.65
	Heavy	0.12
	Total	1.77
In transit:	Resting	0.86
	Light	0.86
	Moderate	0.05
	Heavy	0.0012
	Total	1.77

Source: Adapted from U.S. EPA, 1990d

Swimming Activities. The Superfund Guidelines state that local climatic conditions (e.g., number of days above a certain temperature) and age of potentially exposed population should be taken into consideration in evaluating how often individuals may be engaged in swimming activities in surface waters (U.S. EPA, 1989a). The Guidelines indicate a default value of 7 days per year based on the national average. EPA Region IV has issued guidance stating that an exposure frequency of 45 days/yr should be used for risk assessments conducted in the southeast region (U.S. EPA, 1991e).

Other Values - ATSDR recommends using as standard reference values 70 years as the average life span, 30 years as the national upper bound for the length of time at one residence (90th percentile), and 9 years as the national median for the length of time at one residence (50th percentile).

Information from the U.S. Bureau of Census housing surveys was used by Israeli and Nelson (1992) to estimate residential occupancy periods for different types of housing (owned vs. rented). Based on this data, Finley et al. (1994) compiled distribution percentiles and cumulative density functions for residential occupancy periods. An abbreviated version of the selected distribution percentiles is shown in Table 2-6. Additional information on residential occupancy can also be found in more recent Bureau of Census surveys (U.S. Bureau of Census, 1994).

Table 2-6. Residential Occupancy Period (yr) - Selected Distribution Percentiles

Housing Type	Percentile				
	10th	50th	90th	95th	99th
All houses	0.2	1.4	12.9	23.1	60.5
Renters	0.2	1.2	5.2	8.0	57.6
Owners	0.56	5.2	32.0	41.1	64.2
Urban	0.2	1.4	10.9	21.7	60.3
Rural	0.48	3.3	21.7	32.3	62.4
Farms	0.96	10.0	48.3	58.4	67.7

Source: Finley et al., 1994 (based on data of Israeli and Nelson, 1992).

Using the methodology of Johnson and Capel (1992), Finley et al. (1994) characterized the cumulative distribution of duration of time spent in the residence of birth. Selected percentiles for this distribution are: 0.6 years at the 10th percentile; 2.9 years at the 50th

percentile; 9.7 yr at the 90th percentile; 13.0 years at the 95th percentile; and 21.0 years at the 99th percentile.

Using Bureau of Labor Statistics, Finley et al. (1994) characterized the cumulative distribution of duration of time spent in one job. Selected percentiles for this distribution are: 1.0 years at the 10th percentile; 3.8 years at the 50th percentile; 19.0 years at the 90th percentile; 29.0 years at the 95th percentile; and 30 years at the 100th percentile (maximum).

Jenkins et al. (1992) evaluated daily time-use patterns in families living in California. They reported that 62% of the time was spent indoors at home, 25% indoors not at home, 5% outdoors, and 8% in transit. Time-use data complied by Chapin (1974) and Szalai (1972) as summarized by Boström et al. (1994) are shown in Table 2-7.

Table 2-7. Time-Use Data (hr/day)

Activity	Chapin Study	Szalai Study
Indoors:		
at home	16.03	16.75
at work	4.61	4.03
other things	1.31	1.63
Outdoors:		
at home	0.27	0.23
other things	0.27	0.12
In transit:	1.16	1.25

Source: Boström et al., 1994

2.4.3.4 Inhalation Rates

Estimates of exposure resulting from inhalation of vapors or particulate matter are a function of inhalation rate and exposure duration, as well as the contaminant concentration in air and attached to the particulate matter. The inhalation rate is dependent on the age of the individuals in the population exposed as well as on the activity level of the individuals during the exposure.

EPA Default Values - Data compiled by U.S. EPA (1985a) and listed in the Exposure Factors Handbook identify mean inhalation rates by age, sex and activity level (Table 2-8). For Superfund risk assessments in which specific activity patterns are not known, EPA is currently recommending using 20 m^3/day as a reasonable upper bound value for segments of the populations expected to spend most of their time at home (i.e., housewives, household workers, retired people and unemployed workers (U.S. EPA, 1991b). The 20 m^3/day value is based on 8 hr/day of rest and 16 hr/day of light activity (U.S. EPA, 1993a). EPA Region III uses an age-adjusted inhalation rate of 11.66 m^3/day (12 m^3/day for children and 20 m^3/day for adults) to calculate risk-based concentrations which can be used to derive screening levels (U.S. EPA, 1994i).

Table 2-8. Inhalation Rates (m^3/hr)

Activity	Adult Men	Adult Women	Average Adult	6-Yr-Old	10-Yr-Old
Resting	0.7	0.3	0.5	0.4	0.4
Light activity	0.8	0.5	0.6	0.8	1.0
Moderate activity	2.5	1.6	2.1	2.0	3.2
Heavy activity	4.8	2.9	3.9	2.4	4.2

Source: U.S. EPA, 1990d

For calculating exposures resulting from inhalation of contaminants volatilizing from water within the home, EPA uses as default values an indoor inhalation rate of 15 m^3/day and a volatilization factor of 0.5 L/m^3 (U.S. EPA, 1991c).

For exposure situations in which the distribution of the activity patterns is known, the values listed in Table 2-8 should be used together with the appropriate exposure durations as discussed in Section 2.4.3.3.

Other Values - ICRP reported representative inhalation rates of 23 m^3/day for adult males, 21 m^3/day for adult females, 15 m^3/day for a 10-yr-old child, 3.8 m^3/day for a 1-yr-old infant, and 0.8 m^3/day for a newborn infant (ICRP, 1981). ATSDR recommends using these as default values in exposure assessments (ATSDR, 1990). The state of Massachusetts uses 20 m^3/day for adults and 10 m^3/day for children in its risk assessment methodology (MDEQE, 1989).

According to Layton (1993), long-term inhalation rates are controlled primarily by the amount of oxygen an individual consumes during metabolic conversion of food nutrients to energy. Inhalation rates can therefore be expressed as a function of the distribution of the

metabolic oxygen requirements associated with average daily energy expenditures for individuals of various ages. Based on this approach, Finley et al. (1994) estimated distribution percentiles and cumulative density functions for age-specific inhalation rates. An abbreviated version of the selected distribution percentiles is shown in Table 2-9.

Table 2-9. Selected Distribution Percentiles for Inhalation Rates (m^3/day)

Age (yr)	Percentile				
	10th	50th	90th	95th	99th
<3	3.6	4.7	6.2	6.7	7.8
3-10	6.5	8.4	10.9	11.8	13.8
10-18	9.8	13.1	17.7	19.3	22.5
18-30	11.3	14.8	19.5	21.0	24.6
30-60	9.1	11.8	15.4	16.7	19.2
>60	9.2	11.9	15.6	16.7	19.6

Source: Finley et al., 1994 (based on data of Layton et al., 1993).

2.4.3.5 Ingestion of Drinking Water and Water-Based Beverages

EPA Default Values - The standard value previously used by EPA for drinking water consumption is 2 liters per day for adults (U.S. EPA, 1989a). Data complied in the Exposure Factors Handbook indicate that an intake of 2 liters per day corresponds to the 84th percentile and that 1.4 liters per day is closer to an average value (U.S. EPA, 1990d). These values have been recommended for use at Superfund sites (U.S. EPA, 1993a). The Superfund guidelines suggest that the age-specific drinking water consumption rates listed in the Exposure Factors Handbook be used for children (see Table 2-10). EPA Region III uses an age-adjusted intake rate of 1.09 L/day (1 L/day for children 1-6 and 2 L/day for ages 7-30) to calculate risk-based concentrations which can be used as screening levels (U.S. EPA, 1994i). A standard reference value of 1 liter per day for children of 10 kg body weight or less is used in the derivation of Ambient Water Quality Criteria and Drinking Water Health Advisories (U.S. EPA, 1994a).

For residential exposure scenarios, EPA has estimated that 75 to 100% of the total daily intake of tap water and tap water-based drinks would occur in the home (U.S. EPA, 1990d). For occupational exposures, EPA suggests that until more data becomes available,

it should be assumed that half of an individual's daily water intake occurs at work, unless site-specific data (outdoor workers in hot weather) suggest a higher intake (U.S. EPA, 1991b).

Table 2-10. Fluid Consumption Rates for Infants and Children (L/day)

Age group	Total fluids	Water and water-based foods	Tapwater	Reference
6-11 mo	0.689	0.201	—	Pennington, 1983
< 1 yr	-	0.307 (±0.089)	0.170 (±0.064)	U.S. EPA, 1984a
1-4 yr	-	0.743 (±0.043)	0.4346 (±0.031)	U.S. EPA, 1984a
2 yr	0.930	0.499	—	Pennington, 1983
5-9 yr	-	0.861 (±0.036)	0.521 (± 0.026)	U.S. EPA, 1984a
5-14 yr	1.0-1.2	~0.580	~0.200	ICRP, 1981
5-14 yr	1.31-1.67	0.54-0.79		ICRP, 1981
10-14 yr	-	1.025 (±0.034)	0.620 (± 0.024)	U.S. EPA, 1984a

Source: U.S. EPA, 1990d; standard error of the mean is shown in parenthesis.

Other Values - ATSDR recommends using as default values 2 L/day for adults and 1 L/day for children (ATSDR, 1990). The U.S. Army uses a value of 1.6 L/day for adults (USABRDL, 1989).

The literature data on which EPA revised its estimate of adult drinking water ingestion rates are presented in Table 2-11. Data on tap water ingestion rates in children are shown in Table 2-10. A rate less than 0.2 L/day has been reported for children under 1 year of age, and rates of 0.2-0.6 L/day have been reported for older children.

Ershow and Cantor (1989) utilized the results of the 1977-78 Nationwide Food Survey of the Department of Agriculture to conduct a statistical analysis of total fluid and tap water intake for different age groups. From this analysis, Finley et al. (1994) developed distribution percentiles and cumulative density functions for age-specific tap water intake rates. An abbreviated version of the selected distribution percentiles is shown in Table 2-12. Finley et al. (1994) note that these percentiles should probably be regarded as an overestimate of the intake rates for the upper tail of the distribution because the original data were based on short-term (3-day) sampling.

Table 2-11. Adult Drinking Water Consumption Rates (L/day)			
Average	Range	90th Percentile	Reference
1.63[a]	-	-	NAS, 1977
1.39	≤0.80-≥1.96	2.0[b]	Cantor et al., 1987
1.25	0.26-2.80	1.90	Gillies and Paulin, 1983
1.20	-	-	Pennington, 1983
1.53	1.24-1.73	1.68[b]	U.S. EPA, 1984a

Source: U.S. EPA, 1990d
[a] Calculated
[b] Estimated

Table 2-12. Selected Distribution Percentiles for Tapwater Intake (L/day)					
	Percentile				
Age (yr)	10th	50th	90th	95th	99th
<1	0.0	0.24	0.65	0.78	1.1
3-11	0.29	0.67	1.3	1.5	2.0
11-18	0.35	0.87	1.7	2.0	2.7
18-65	0.56	1.3	2.3	2.7	3.8
>65	0.75	1.4	2.3	2.6	3.3

Source: Finley et al., 1994 (based on data of Ershow and Cantor, 1989)

2.4.3.6 Ingestion of Soil

Total exposure to a contaminant through ingestion of soil is a function of age-specific ingestion rates, exposure duration, and the fraction of soil ingested from the contaminated source. Climate, access and time spent outdoors, and proximity to open undeveloped areas will affect exposure levels. The age group of 1-6 years old is given special attention by EPA because of the higher soil intake per unit body weight that is expected to occur in this group (U.S. EPA, 1990d, 1993a).

EPA states that default values for exposure frequency and duration may not always be appropriate in estimating exposures through soil ingestion (U.S. EPA, 1993a). Site-specific factors, such as the unique uses of the site (i.e., child care facility, farms), or regional

differences in climate may substantially influence these parameters. Other factors that can affect the estimates of exposure are the fraction of soil ingested that comes from the contaminated source, and characteristics of the contaminated soil that determine the rates of desorption of contaminant from the soil following ingestion. Default values are not available for these variables (U.S. EPA, 1993a).

EPA Default Values - In the Superfund Guidelines, recommended default values for soil ingestion are 200 mg/day for children 1-6 years old and 100 mg/day for children older than 6 and adults (U.S. EPA, 1989a). Using the data of Binder et al. (1986) and Clausing et al. (1987), EPA estimated that 200 mg/day would be a typical soil consumption rate for children 1-6 years old, and 800 mg/day would be a "reasonable worst case value" (U.S. EPA, 1990d).

Soil/dust ingestion values of 200 mg/day for young children and 100 mg/day for older children and adults (only for those adults not engaged in activities in which contact with soil or dust is excessive) have been reproposed as the default values for reasonable maximum exposures (U.S. EPA, 1993a). For individuals, such as gardeners or construction workers, who routinely come into contact with soil, the proposed default is 480 mg/day as a reasonable maximum exposure (U.S. EPA, 1993a). The corresponding proposed default values for what would be considered typical levels of soil ingestion (central tendency) are: 100 mg/day for children 1-6 years old and 50 mg/day for adults not engaged in soil contact intensive activities (a CT default value for adults engaged in soil intensive activities was not proposed). For residential exposure scenarios, EPA has proposed that overall soil ingestion be calculated on the basis of time-weighted average exposure; i.e., for a 9-yr average period residency at one location, 2 years at 100 mg/day (children) and 7 years at 50 mg/day (adults), and for a reasonable maximum 30-yr period of exposure, 6 years at 200 mg/day (children) and 24 years at 100 mg/day (adults) (U.S. EPA, 1993a). In both cases the default exposure frequency is 350 days per year for residential scenarios (U.S. EPA, 1993a). This time-weighted average method has been used in EPA's proposed Soil Screening Level methodology, where a default value of 114 mg/day is identified as a reasonable maximum value for 30-year occupancy at a single location (U.S. EPA, 1993b).

Other Values - Standard soil ingestion rates used by various state and federal agencies in exposure assessments are shown in Table 2-13.

In developing a soil contact exposure scenario to be used to calculate an Applied Action Level (AAL) for the State of California's Site Mitigation Decision Tree Manual, Sedman (1989) evaluated the available literature, and derived a time-weighted average daily soil ingestion rate of 0.15 g/day. This value was based on a 70-year lifetime exposure and on estimates of age-specific ingestion rates. From the results of five soil ingestion studies, Sedman (1989) estimated that ingestion rates for children ranged from 0.04 to 0.64 g/day

(arithmetic mean was 0.33 g/day with a standard deviation of 0.26 g/day). The ingestion rate for children 1-3 years old was 0.59 g/day and that for children 3-19 years old was expressed by the function $0.74\ e^{-0112x}$, where x is the age of the child in years. The average ingestion rate for adults was estimated to be 0.10 g/day, and the average over a lifetime was 0.15 g/day.

Table 2-13. Soil Ingestion Rates (mg/day) Used by State and Federal Agencies

Age Group	EPA Superfund[a]	ATSDR[b]	U.S. Army[c]	Mass.[d]	Wash.[e]	Mich.[f]	Calif.[g]
0-1		50		50			
1-6	100 (CT) 200 (RME)	100	820	100			
7-11	100	50	820	50			
adult	50 (CT) 100 (RME)	25-50	74	50			
1-31	114 (RME)						
lifetime					200[e]	90	150
occupational	480 (RME)						

[a] CT = central tendency; RME = reasonable maximum exposure
[b] ASTDR, 1990
[c] Small, 1989
[d] MDEQE, 1989
[e] WDOE, 1990, value show is for establishing cleanup levels
[f] MCEQ, 1990
[g] CAPCOA, 1987

In preliminary mass balance studies conducted by Calabrese et al. (1990; 1991), a soil ingestion rate of about 50 mg/day was predicted for adults; however, in a later report the same investigators noted that these estimates were below the detection capability as determined by a mathematical model predicting soil ingestion recovery values (Calabrese and Stanek, 1991; Stanek and Calabrese, 1991). Finley et al. (1994) state that currently available data are inadequate for estimating adult soil ingestion rates, and the same opinion was expressed in EPA's Exposure Factors Handbook (U.S. EPA, 1990d).

Soil ingestion rates in children have been studied by Calabrese and coworkers using soil trace element methodologies (Calabrese et al., 1989, Stanek et al., 1991; Calabrese and Stanek, 1991 and 1995; Stanek and Calabrese, 1995). Sixty-four children 1-4 years old participated in the 2-week study that took place in September and October in Amherst, MA.

Six trace elements were used in the studies; aluminum, silicon, titanium, vanadium, yttrium, and zirconium. Early results indicated poor intertracer consistency. Mass balance studies identified the principal sources of positive and negative errors in the estimates. After applying correction factors, the mean soil ingestion rates in children ranged from 97 to 208 mg/day for the six tracers (Calabrese and Stanek, 1995). The data indicated that aluminum, silicon, and yttrium provided the most reliable estimates. The data were also used to calculate a single, median estimate of soil ingestion for each subject for each day (Stanek and Calabrese, 1995). The median soil ingestion rate was 13 mg/day or less for 50% of the children and 138 mg/day or less for 95% of the children. Mean soil ingestion estimates (for up to an 8-day period) were 45 mg/day or less for 50% of the children and 208 mg/day or less for 95%.

In general, current data do not appear to be adequate for deriving reliable distributions for long-term soil ingestion by children; however, several estimates have been made. Stanek and Calabrese (1995) used the modified Amherst study data to develop a cumulative distribution profile for a one year time period assuming that soil ingestion followed a log normal distribution. As averaged over 365 days, the median soil ingestion rate was 75 mg/day, while the upper 95% was 1751 mg/day. These estimates were limited by the small number of samples and the short period of time (2 weeks) over which the samples were taken. Finley et al. (1994) used the zirconium data from the Calabrese studies to estimate distribution percentiles and a cumulative density function for children age 1-4 yr. Selected distribution percentiles were 0 for the 10th percentile, 16 mg/day for the 50th percentile, 67.0 mg/day for the 90th percentile, and 110 mg/day for the 95th percentile. Finley et al. (1994) reported that these percentiles should probably be regarded as overestimates because the original data were based on short-term sampling. It should also be noted that Calabrese and Stanek (1995) have recently determined that zirconium did not provide a reliable estimate of soil ingestion for children. In evaluating the use of Monte Carlo simulations in health risk assessments, Thompson et al. (1992) used a lognormal distribution for soil ingestion. The reported mean value (as derived from the "underlying normal distribution") was 3.44 mg/day (standard deviation 0.80).

EPA (1990d) cited unpublished data from Calabrese et al. (1987) concerning soil ingestion by children with a "high tendency to ingest soil". These values are: 1000 mg/day for children 9-18 months old, 10,000 mg/day for children 1.5-3.5 years old, 1000 mg/day for children 3.5-5.0 years old, and 100 mg/day for children 5-18 years old.

2.4.3.7 Ingestion of Homegrown Fruits and Vegetables

<u>EPA Default Values</u> - If ingestion of homegrown fruits or vegetables is considered to be a likely exposure pathway, EPA recommends that local surveys be conducted to determine consumption rates or determine productivity levels for local gardeners and derive consumption

rates by dividing productivity by the number of consumers. Using national survey data from the USDA (1980), and information derived from 3-day dietary records (Pao et al., 1982), EPA estimated consumption rates for homegrown fruits and vegetables (U.S. EPA, 1991b, 1993a). Recently, however, EPA conducted an analysis of USDA 1987-1988 data which will substantially change the estimates of ingestion rates for homegrown fruits and vegetables. This new data will appear in the revised Exposure Factors Handbook which is scheduled for release by EPA in the near future.

Other Values - ASTDR recommends calculating exposure by determining the concentration of the contaminant in each food group consumed, and using information on the amount of that food group eaten each day and the percentage of that food group that is home-grown (PRF, 1980) to derive an intake value for each food group and then determine total exposure by summing intakes for all groups.

2.4.3.8 Ingestion of Homegrown Beef and Dairy Products

EPA Default Values - If this is considered to be a likely exposure pathway (i.e., for farm families), EPA recommends conducting local surveys to determine consumption rates. Using national survey data, EPA estimated that consumption of homegrown beef and dairy products would typically be about 44 g/day for beef and 160 g/day for dairy products (U.S. EPA, 1991b). Reasonable maximum estimates of 75 g/day for beef and 300 g/day for dairy products were based on the assumption that the percentage of annual consumption that is home grown is 75% for the 90th percentile consumer. EPA recommends that the procedures described in Travis and Arms (1988) be used to estimate organic contaminant concentrations in beef and milk.

2.4.3.9 Ingestion of Locally Caught Fish and Shellfish

EPA Default Values - For estimating the consumption rates of recreationally caught fish, EPA recommends using the averages for the 50th and 90th percentile values from the survey data reported by Puffer (1981) and Pierce et al. (1981) (U.S. EPA, 1990d). These default values are 30 g/day and 140 g/day, respectively. However, these values apply only to large water bodies where widespread contamination is evident. Furthermore, the Puffer (1981) study was conducted in the Los Angeles area and the Pierce study was conducted in Commencement Bay, WA and the results may not be applicable to other areas of the country. For small water bodies or localized areas of contamination, EPA recommends: (1) using local surveys, (2) obtaining productivity data and dividing by the number of consumers, (3) estimating the diet fraction of locally caught fish that is eaten locally and applying this value to the 50th and 90th

percentile consumption data for the large water bodies, or (4) develop standard exposure scenarios assuming the number of fish meals eaten from the area per year and applying a meal size in the range of 100 to 200 g/meal.

For Superfund risk assessments, a value of 145 g is the recommended default for the average amount of fish eaten per meal (U.S. EPA, 1993a). It was suggested that defaults for frequency of exposure (fish meals/year) and estimates of the fraction of fish consumed from contaminated source be based on site-specific data (the average and 90-95th percentile are suggested as the values to use for the CT and RME). If there is evidence that the local population consumes fish from the impacted water body on a regular basis (subsistence fishing, resulting in about 4 8-oz servings per week), then EPA suggests that the 95th percentile, or 132 g/day, be used as the RME (U.S. EPA, 1991b).

Other Values - Standard fish consumption values used by various federal and state agencies are shown in Table 2-14.

Table 2-14. Fish Consumption Values (g/day) Used by State and Federal Agencies

EPA	ATSDR[a]	U.S. Army[b]	Mass.[c]	Wash.[d]	Mich.[e]
30 (CT)	12	6.5	20	30	6.5

Source: Whitmyre et al., 1992a
[a] ASTDR, 1990
[b] USABRDL, 1989
[c] MDEQE, 1989
[d] WDOE, 1990
[e] MCEQ, 1990

Using the results of the 1973-74 National Marine Fisheries Service Study on fish consumption in the U.S., as reported by Rupp et al. (1980), Finley et al. (1994) calculated distribution percentiles and cumulative density functions for fish and shellfish consumption for the general U.S. population. An abbreviated version of the selected distribution percentiles is shown in Table 2-15.

Finley et al. (1994) also present the data of Ebert et al. (1993) and that of Puffer et al. (1981), as reanalyzed by Pierce et al. (1981) for distribution percentiles and cumulative density functions for fish and shellfish consumption for anglers who catch and consume fish from different types of water bodies. An abbreviated version of the selected distribution percentiles is shown in Table 2-16. It should be noted that the Ebert et al. (1993) study focused on recreational freshwater fishermen in the state of Maine, whereas the Pierce et al. (1981) study focused on the general angler population in the Los Angeles area.

Table 2-15. Selected Distribution Percentiles for Fish and Shellfish Consumption (g/day)								
Age (yr)	Percentile							
	50th		90th		95th		100th	
	Fish	Shellfish	Fish	Shellfish	Fish	Shellfish	Fish	Shellfish
1-11	1.1	0	4.1	1.4	8.0	2.8	47.9	12.9
12-18	2.4	0	7.8	1.9	13.8	4.0	48.1	14.1
19-98	2.7	0	10.6	4.0	31.0	8.2	123.1	44.1

Source: Finley et al., 1994 (based on data from Rupp et al., 1980).

Table 2-16. Selected Distribution Percentiles for Fish and Shellfish Consumption (g/day) for Anglers					
Source	Percentile				
	10th	50th	90th	95th	99th
Freshwater[a]:					
Rivers and streams	0.17	0.99	6.1	12.4	49
Lakes and ponds	0.27	1.7	8.5	15.0	53
Total	0.3	2.0	13.0	26.0	73
Marine[b]:					
Total	0.18	1.0	13.3	32.9	154

[a] **Source:** Finley et al., 1994 (based on data of Ebert et al., 1993)
[b] **Source:** Finley et al., 1994 (based on data of Puffer et al., 1981, as evaluated by Pierce et al., 1981)

Paustenbach et al. (1992) used a fish consumption rate of 1.48 g/day in their derivation of guidelines for generic exposure scenarios. They also included as exposure factors the potential loss of contaminant during cooking, and an estimate of the number of days the fish are contaminated.

2.4.3.10 Dermal Contact with Soil and Soil-to-Skin Adherence

EPA Default Values - U.S. EPA (1992b) has suggested that for most soil contact scenarios for adults, the hands, legs, arms, neck and head would be exposed and that the exposure would be equivalent to 25% of the total body surface area. Since the adult total body surface area is about 2.0 m^2 (see Section 2.4.3.2), EPA recommends using default values of 5000 cm^2 (50th percentile) to 5800 cm^2 (95th percentile) for adults.

For Superfund sites, dermal exposure of children 1-6 years old is given special attention because of the higher intake to body weight ratio that is expected for this age group. EPA recommends that the factor of 25% of the total body surface area be used as the default value for the amount of exposed skin in children (U.S. EPA, 1989a). U.S. EPA (1985a) has estimated age-specific body surface areas and exposed surface areas for children and adults (Table 2-17).

The EPA default values for soil-to-skin adherence are 0.2 and 1.0 mg-soil/cm^2-skin for the "reasonable" and "upper bound values" (U.S. EPA, 1992a; 1992c).

Other Values - Using the data presented by EPA (1985a), ATSDR (1990) recommends using the following default values: 1050 cm^2 skin area exposed in children 0-1 years old; 2625 cm^2 for children 1-11 years old; 4300 cm^2 for children 12-17 years old; and 4700 cm^2 for adults 18-70 years old. ATSDR (1990) also recommends using a soil-to-skin adherence default value of 2.0 mg/cm^2.

Table 2-17. Body Surface Area and Exposed Surface Areas (cm^2)*

Age (yr)	Total surface	Exposed surface
2 - 3	6,030	2,050
3 - 4	6,640	2,258
4 - 5	7,310	2,485
5 - 6	7,930	2,379
6 - 7	8,660	2,598
7 - 8	9,360	2,808
8 - 9	10,000	3,000
9 - 10	10,700	3,210
14 - 15	16,100	4,186
17 - 18	18,000	4,320
18 - 70	19,400	4,656

* **Source:** U.S. EPA, 1985a

Driver et al. (1989) conducted a series of studies to determine the effect of soil type, particle size, and organic content on adherence of soil to hands. They found that adherence was significantly correlated with organic content and particle size. Mean adherence values of 0.58 mg/cm^2 for unsieved soil (five soil types), 0.94 mg/cm^2 for the particle size fraction of < 250 μm, and 1.40 mg/cm^2 for the particle size fraction of < 150 μm. Mean adherence was 0.58 mg/cm^2 for soils with 19% organic matter, but only 0.78 mg/cm^2 for soils having 0.77% organic matter.

In developing a soil contact exposure scenario to be used to calculate an Applied Action Level (AAL) for the State of California's Site Mitigation Decision Tree Manual, Sedman (1989) evaluated the available literature, and derived a time-weighted average daily dermal soil exposure value of 0.45 g/day. This value was based on a 70-year lifetime exposure and on estimates of age-specific soil-to-skin loading rates; i.e., 0.5 g/cm^2 for children 1-3 years old; 0.74 g/cm^2 x $e^{-0.112x}$, where x equals the age in years for children 3-18 years old; and 0.74 g/cm^2 x $e^{-0.112x}$, where x equals 18, for adults.

Finley et al. (1994) reviewed the available data and concluded that soil-to-skin adherence does not significantly vary with age, sex, soil type, particle size, or indoor or outdoor exposure. They derived a distribution with the following percentiles: 0.03 mg-soil/cm^2 at the 10th percentile; 0.25 mg-soil/cm^2 at the 50th percentile; 1.2 mg-soil/cm^2 at the 90th percentile, and 1.7 mg-soil/cm^2 at the 95th percentile.

Kissel et al. (1996) measured soil loading on skin surfaces of volunteers before and after normal occupational and recreational activities. Hands, forearms, lower legs, faces, and feet were assayed. Hand loadings ranged from about 0.001 to 140 mg/cm^2. Hand loadings of 0.2-1.0 mg/cm^2 resulted from high impact activities such and rugby and farming; hand loadings less than 0.2 mg/cm^2 were associated with soccer players and groundskeepers. Children playing in the mud on the shore of a lake generated geometric mean loadings in excess of 1 mg/cm^2 on hands arms legs, and feet, with a maximum value of 140 mg/cm^2 for hands. Kissel et al. (1996) considered the data adequate to define point estimates of hand loadings for specific classes of activity; i.e., 0.01, 0.1, 1.0, and 10 mg/cm^2 for background, low, moderate and high contact activities, respectively. Average loading on the hands was typically larger than that for the arms or legs for a given exposure scenario.

2.5 INTEGRATED EXPOSURE/DOSE ANALYSIS

The exposure scenarios described in Section 2.4 have several functions in risk assessments; they are a means to quantify exposure and dose; they are a way to develop risk descriptors; and they are a tool for evaluating remediation options (U.S. EPA, 1992a). The

procedure for estimating exposure and dose and expressing the results in terms of risk descriptors is presented in the following sections.

2.5.1 Estimating Exposure and Dose from Contaminant Concentrations

Exposure results from the contact of an organism with environmental media (air, water, food, soil) containing the contaminant. Exposure, as measured by the intensity and duration of contact at the external boundary of the organism, can be represented by a time-dependent profile of the environmental concentration at the point of contact. The area under the curve of the profile is the magnitude of the exposure in concentration-time units. The biologically significant measure of exposure, the intake or dose, is the amount of the contaminant that crosses the external boundary of the organism. Substances cross the external boundaries of an organism primarily through inhalation, ingestion, or absorption through the skin. A brief summary of the concepts of exposure and dose, as described in the EPA Exposure Assessment Guidelines is presented in the following sections.

2.5.1.1 Ingestion and Inhalation Doses

For the ingestion and inhalation pathways, the assumption is made that the substance is homogeneously distributed through the environmental medium (e.g., air, water) and that the intake of the chemical into the body is proportional to the intake of the medium. Therefore, the dose is the integration of the chemical intake rate (i.e., the exposure concentration times the intake rate of the medium) over time. In cases where the actual contact or exposure time is short and intermittent, such as in the ingestion of food or drinking water, the intake rate can be expressed in terms of the frequency of events times the intake per event, and the total dose can be expressed as the sum of the doses received during each event. If the assumption is made that the exposure concentration and the intake of the medium are nearly constant during each event, the resulting dose can be expressed as:

$$D = \overline{C} \cdot \overline{IR} \cdot ED \qquad (2\text{-}1)$$

where:

D = dose (e.g., mg)
ED = exposure duration (or frequency) (e.g., days)
$\overline{C}$ = concentration (e.g., mg/L for amount in water)
$\overline{IR}$ = intake rate (e.g., L/day for drinking water)

Doses are usually expressed in terms of the rate of intake of the substance per unit time (e.g., mg/day) per unit body weight (e.g., mg/kg body weight/day). Therefore, for the above equation, the average daily dose (ADD) is:

$$ADD = \frac{\overline{C} \cdot \overline{IR} \cdot ED}{BW \cdot AT} \qquad (2\text{-}2)$$

where:

ADD = average daily dose (mg/kg/day)
ED = exposure duration (or frequency) (days)
$\overline{C}$ = concentration (e.g., mg/L for amount in water)
$\overline{IR}$ = intake rate (e.g., L/day for drinking water)
BW = body weight (kg)
AT = time period over which the dose is averaged (days)

The average daily dose presented above is a measure of the potential or administered dose (ADD_{adm}) at the point of contact with the exposed individual. It may or may not be equivalent to the "applied dose" (ADD_{app}), the dose available for absorption at the tissue interface (i.e., at the surface of the respiratory or gastrointestinal tract), and it may or may not be equivalent to the "internal dose" ($ADD_{internal}$), the dose that is absorbed and is available for interaction with biological significant receptors. Theoretically, the internal dose can be estimated from the concentration of the chemical in the gastrointestinal or respiratory tract, the internal surface area, and the permeability coefficient for the specific tissue; however, this information is not usually available. Therefore, EPA recommends that it be assumed that the applied dose is equal to the administered (potential) dose (i.e., that all the chemical ingested in food or drinking water or all that is inhaled touches an absorption barrier inside a person). If this assumption is made, then the following equation can be used to estimate the internal dose:

$$D_{internal} = \overline{C} \cdot \overline{IR} \cdot ED \cdot AF \qquad (2\text{-}3)$$

where:

AF = absorption fraction (units of mass absorbed per units of mass applied)

And the average daily internal dose ($ADD_{internal}$) becomes:

$$ADD_{internal} = \frac{\overline{C} \cdot \overline{IR} \cdot ED \cdot AF}{BW \cdot AT} \qquad (2\text{-}4)$$

Rates of absorption may differ between individuals, populations and species, resulting in different internal doses for the same administered dose. Because information on absorption factors is limited; EPA has in the past routinely used the administered dose and not the internal dose in deriving toxicity values such as oral Reference Doses (RfD) (although the human equivalent dose might be adjusted if experimental data indicate a significant difference in absorption between the test species and humans). Because the primary objective in risk assessments is to compare exposures to such toxicity values, administered doses are normally used in oral exposure assessments. However, administered doses must be adjusted to internal or absorbed doses for comparison with dermal exposures (see below) and to calculate total dose for all exposure pathways. This is done by using an experimentally derived value or a default value for gastrointestinal absorption. Appendix A of the Superfund Guidelines (U.S. EPA, 1989a) states that in the absence of specific data, an assumption of 5% oral absorption would be a relatively conservative assumption for metals; however, defaults for other inorganic substances and for organic compounds are not given. EPA Region IV has proposed the following defaults: 80% for volatile organic chemicals, 50% for semi-volatile organic chemicals, and 20% for inorganics (U.S. EPA, Region IV, undated).

Reference toxicity values for inhalation exposures are expressed in terms of Reference Concentrations (RfC). For Superfund risk assessments, the RfC is converted to an equivalent RfD by using certain default assumptions (i.e, inhalation rate of 20 m^3/day and a standard body weight of 70 kg), and, in the absence of any specific data, a pulmonary absorption factor of 100% is assumed. In an inhalation exposure assessment, the same assumptions must be used for a valid comparison with the inhalation RfD. However, if the exposure assessment focuses on a population that is substantially different in inhalation rate or body weight, such as children, then comparison with the inhalation RfD may not be appropriate. There is no EPA guidance as to whether in such cases the inhalation RfD should be adjusted using different default parameters.

2.5.1.2 Dermal Doses

Dermal exposures differ from ingestion and inhalation exposures in that the exposure is normally expressed in terms of the internal or absorbed dose, whereas, oral or inhalation

exposures are expressed in terms of an administered dose. An internal dose resulting from absorption of a chemical through the skin can be calculated in two ways depending on the type of medium considered, water or soil.

Dermal Absorption from Water - When a chemical is being absorbed from water following partial immersion of the body, uptake of the chemical is a function of the concentration of the chemical in the water, the surface area of the exposed skin, the permeability coefficient of the skin, and the time of contact. Assuming intermittent short-term exposures with chemical concentration in the water remaining relatively constant, the equation for internal dose can be simplified as:

$$D_{internal} = \overline{C} \cdot K_p \cdot \overline{SA} \cdot ED \qquad (2\text{-}5)$$

where:

$D_{internal}$ = internal dose (mg)
$\overline{C}$ = concentration (mg/L in water)
K_p = permeability coefficient (cm^2/hr)
ED = exposure duration (hr)
$\overline{SA}$ = average surface area exposed (cm^2).

The average daily dose ($ADD_{internal}$), normalized to body weight, can then be expressed as:

$$ADD_{internal} = \frac{\overline{C} \cdot K_p \cdot \overline{SA} \cdot ED}{BW \cdot AT} \qquad (2\text{-}6)$$

where:

ADD = average daily dose (mg/kg/day)
BW = body weight (kg)
AT = averaging time (days)

Dermal Absorption from Soil - For a contaminant in soil, uptake through the skin is a function of chemical concentration, skin surface area exposed, soil-to-skin adherence (or soil loading) factor, and exposure duration:

$$D_{pot} = \overline{C} \cdot F_{adh} \cdot \overline{SA} \cdot ED \qquad (2\text{-}7)$$

where:

D_{pot} = potential dose (mg/day)
$\overline{C}$ = concentration (mg/kg soil)
F_{adh} = soil-to-skin adherence factor (soil loading) (mg/cm^2)
$\overline{SA}$ = average surface area exposed (cm^2)
ED = exposure duration (days)

The soil-to-skin adherence factor (F_{adh}) is the amount of soil applied to and adhering to the skin based on a unit surface area. The type of soil at a given site determines F_{adh}. In the absence of specific data, EPA (1989a) had originally recommended using the following default values: 1.45 mg/cm^2 for commercial potting soil and 2.77 mg/cm^2 for kaolin clay. EPA (1992a) recently reported that "a range of values from 0.2 mg/cm^2 to 1.5 mg/cm^2 per event appear possible". According to the EPA, because "this range is derived from hand measurements only, it may overestimate average adherence for the entire exposed skin area. Thus, the lower end of this range (0.2 mg/cm^2) may be the best value to represent an average over all exposed skin and 1 mg/cm^2 may be a reasonable upper value" (U.S. EPA, 1992a; 1992c).

For dermal exposures, the internal or absorbed dose is a function of the amount of chemical actually touching the skin (applied dose), the rate of absorption through the skin, and the time of contact. If integrated over time, the internal dose can be expressed in terms of the applied dose (D_{app}) and an absorption factor:

$$D_{internal} = D_{app} \cdot AF \qquad (2\text{-}8)$$

where:

AF = absorption fraction (units of mass absorbed per units of mass applied)

If one assumes that all of the chemical contained in the bulk material eventually comes in contact with the skin then D_{app} equals D_{pot}, and:

$$D_{internal} = D_{pot} \cdot AF \qquad (2\text{-}9)$$

and from Equation 2-7:

$$ADD_{internal} = \frac{\overline{C} \cdot F_{adh} \cdot \overline{SA} \cdot ED \cdot AF}{BW \cdot AT} \qquad (2\text{-}10)$$

where:

ADD = average daily dose (mg/kg/day)
BW = body weight (kg)
AT = averaging time (days)

EPA notes that the estimate of the internal dose derived in this manner may be unreliable, especially if the exposure is to large amounts of bulk material, in which case the applied dose may be much less than the potential dose. Many uncertainties may also exist in estimating the absorption fraction. Experimentally derived AF values are often based on potential and not applied doses, and are usually reported as the fraction absorbed after a certain time which may or may not be sufficiently long to define steady-state conditions. An AF based on a single data point, or for a short duration or one which was obtained in an experiment in which the exposure conditions were very different from the conditions being considered for an assessment should be viewed as having a low level of certainty, and this would also apply to the estimated $ADD_{internal}$.

A default value for the skin absorption factor (AF) has not been adopted agency-wide by EPA; however, EPA Region IV has recommended using a default value of 1.0% for organic compounds and 0.1% for inorganic compounds; these defaults take into account the soil matrix effect (U.S. EPA, 1992c). Based on the studies of Ryan et al. (1987), the Commonwealth of Massachusetts recommends the following defaults for dermal absorption: 10-25% for VOCs; 1-10% for SVOCs; 1-10% for pesticides; and 0.1-1% for inorganics (MDEQE, 1989).

2.5.2 Dose Equations for Specific Exposure Pathways

EPA has developed standard equations for calculating doses associated with specific exposure pathways (U.S. EPA, 1989a, 1990d). For computing cancer risk, the exposure is averaged over lifetime. For noncancer effects, the exposure is averaged over a specific period

of time such as time of residency time at one location or length of employment at one location. The following sections summarize EPA's standard exposure/dose equations.

2.5.2.1 Ingestion of drinking water

The Superfund Guidelines (U.S. EPA, 1989a) provide the following equation for calculating exposures resulting from the ingestion of contaminated tap water and beverages made from tap water. The equation is based on the assumption that all tap water consumed comes from one contaminated source.

$$Intake = \frac{CW \times IR \times EF \times ED}{BW \times AT} \quad (2\text{-}11)$$

where:

Intake = mg/kg-day
CW = Chemical concentration in water (mg/L)
IR = Ingestion rate [for adults 1.4 L/day (average) and 2.0 L/day (90th percentile); age-specific values for other age groups]
EF = Exposure frequency (365 days/yr for residents)
ED = Exposure duration (residential, 9 years, median; 30 years, 90th percentile)
BW = Body weight (70 kg for adults, age-specific values for other age groups)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

A similar equation is presented in the Exposure Factors Handbook (U.S. EPA, 1990d); however, in this case an allowance is made for the fraction of tap water that is ingested at home, and the lifetime is set at 75 yr:

$$LADE = \frac{CR \times C \times ED \times DF}{BW \times LT \times (365\ days/yr)} \quad (2\text{-}12)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
CR = Ingestion rate (1.4 L/day, avg.; 1.4-2.0 L/day, range)
C = Chemical concentration in water (mg/L)
ED = Exposure duration (avg., 365 days/year for 9 yr = 3,285 days; range = 3,285-10,950 days)
DF = Diet fraction (fraction consumed at home from contaminated source; 0.75, average; 0.75-1.0, range)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

The exposure equation used to derive a Preliminary Remediation Goal (PRG) as described in Part B of the Superfund manual utilizes a standard drinking water ingestion rate of 2 L/day; an exposure duration of 350 days/year for 30 years and an averaging time of 30 years for noncarcinogens and 70 years for carcinogens (U.S. EPA, 1991c).

2.5.2.2 Ingestion of homegrown fruits and vegetables

The Superfund Guidelines (U.S. EPA, 1989a) provide the following equation for estimating exposure/dose resulting from the ingestion of contaminated homegrown fruits and vegetables:

$$Intake = \frac{CF \times IR \times FI \times EF \times ED}{BW \times AT} \qquad (2\text{-}13)$$

where:

Intake = mg/kg-day
CF = Chemical concentration in food (mg/g), site-specific
IR = Ingestion rate (kg/meal) for specific fruits and vegetables (see Pao et al., 1982)
FI = Fraction ingested from contaminated source (site-specific)
EF = Exposure frequency (meals per year)
ED = Exposure duration (residential: 9 years, median; 30 years, 90th percentile)
BW = Body weight (70 kg for adults)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

The Guidelines do not give any default values for IR, FI, or EF, but indicate that these values should be determined by location and size of contaminated area and anticipated usage pattern.

A similar equation is provided in the Exposure Factors Handbook (U.S. EPA, 1990d):

$$LADE = \frac{CR \times C \times ED}{BW \times LT \times (365\ days/yr)} \qquad (2\text{-}14)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
C = Chemical concentration in food (mg/g)
CR = Rate of ingestion of homegrown produce: vegetables - average, 50 g/day; range, 50-80 g/day fruit - average 28 g/day, range 28-42 g/day
ED = Exposure duration (650 days, average; 650-5,500 days, range)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

In this equation, CR is the ingestion rate of homegrown fruits or vegetables, and therefore incorporates both the CF and FI factors in Equation 2-13. The default values for CR are based on national survey results indicating total intake rates of 140 g fruit/day and 200 g vegetables/day. The default values for the dietary fractions coming from the contaminated source (i.e., homegrown) were 20% (average) and 30% (reasonable worst case) for fruits and 25% (average) and 40% (reasonable worst case) for vegetable consumption.

Finley et al. (1993) have derived an equation for estimating exposure resulting from the ingestion of garden vegetables grown on soil irrigated with contaminated tapwater (or well water). The exposure is a function of the contaminant concentration in the tap or well water and the soil-water and plant-soil partition coefficients:

$$Dose = \frac{C \times SW \times PS \times IR \times FH \times CF \times EF \times ED}{BW \times AT} \qquad (2\text{-}15)$$

where:

Dose = mg/kg-day
C = Chemical concentration in water
SW = Soil-water partition coefficient (Liters/kg)
PS = Plant-soil partition coefficient (unitless)
IR = Ingestion rate (kg/day)
FH = Fraction ingested that is homegrown
CF = Conversion factor (10^{-6} kg/mg)
EF = Exposure frequency (days/year)
ED = Exposure duration (years)
BW = Body weight (kg)
AT = Averaging time (yr)

2.5.2.3 Ingestion of homegrown meat and dairy products

The Superfund Guidelines (U.S. EPA, 1989a) provide the following equation for calculating exposure resulting from the ingestion of contaminated meats, eggs and dairy products:

$$Intake = \frac{CF \times IR \times FI \times EF \times ED}{BW \times AT} \qquad (2\text{-}16)$$

where:

Intake = mg/kg-day
CF = Chemical concentration in food (mg/g), site-specific
IR = Ingestion rate (kg/meal) (see Pao et al. 1982, for specific values):
beef - 0.112 kg/meal (50th percentile), 0.28 kg/meal (95th percentile)
eggs - 0.064 kg/meal (50th percentile), 0.15 kg/meal (95th percentile)
FI = Fraction ingested from contaminated source (site-specific)
EF = Exposure frequency (meals per year)
ED = Exposure duration (median, 9 yr; upper bound, 30 yr)
BW = Body weight (70 kg for adults, age-specific values for other age groups)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

The Guidelines do not give any default values for FI, or EF, but indicate that these values should be determined by location and size of contaminated area and anticipated usage pattern.

The exposure/dose equation given in the Exposure Factors Handbook (U.S. EPA, 1990d) incorporates the IR, FI, and EF parameters into a single parameter, CR. The range of intake and duration values represent typical and reasonable worst case values for rural farm households.

$$LADE = \frac{CR \times C \times ED}{BW \times LT \times (365\ days/yr)} \qquad (2\text{-}17)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
CR = Ingestion of homegrown meat and dairy products:
beef - 44 g/day (average); 44-75 g/day (range)
dairy - 160 g/day (average); 160-300 g/day (range)
C = Contaminant concentration in food (mg/g)
ED = Exposure duration (average, 365 days/yr for 20 yr = 7,300 days; range, 20-40 yr)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

2.5.2.4 Ingestion of recreationally caught fish and shellfish

The Superfund Guidelines (U.S. EPA, 1989a) provide the following equation for calculating exposure resulting from ingestion of contaminated fish or shellfish:

$$Intake = \frac{CF \times IR \times FI \times EF \times ED}{BW \times AT} \quad (2\text{-}18)$$

where:

Intake = mg/kg-day
CF = Chemical concentration in fish (mg/g), site-specific
IR = Ingestion rate (g/day) (see Pao et al. 1982, for specific values)
6.5 g/day, intake average over a year
38 g/day (50th percentile, 3-day average) (Pao et al., 1982)
132 g/day (95th percentile, 3-day average) (Pao et al., 1982)
FI = Fraction ingested from contaminated source (site-specific)
EF = Exposure frequency (meals per year)
ED = Exposure duration (median, 9 yr; upper bound, 30 yr)
BW = Body weight (70 kg for adults, age-specific values for other age groups)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

The Guidelines do not give any default values for FI, or EF, but indicate that these values should be determined anticipated local usage patterns.

A similar equation is given in the Exposure Factors Handbook (U.S. EPA 1990d):

$$LADE = \frac{CR \times C \times ED \times DF}{BW \times LT \times (365\ days/yr)} \quad (2\text{-}19)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
CR = Ingestion rate (average, 30 g/day; range, 30-140 g/day)
C = Chemical concentration in food (mg/g)
ED = Exposure duration (average, 365 days/yr for 9 yr = 3.285 days; range, 9-30 yr)
DF = Diet fraction (fraction consumed at home from contaminated source) (average, 0.20; range, 0.2-0.75)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

2.5.2.5 Ingestion of soil - residential setting - children

The Superfund Guidelines (U.S. EPA, 1989a) provide the following equation for calculating exposure to children resulting from ingestion of contaminated soil:

$$Intake = \frac{CS \times IR \times CF \times FI \times EF \times ED}{BW \times AT} \quad (2\text{-}20)$$

where:

Intake = mg/kg-day
CS = Chemical concentration in soil (mg/kg), site-specific
IR = Ingestion rate (g/day)
200 mg/day, children 1-6 yr old
100 mg/day, children >6 and adults
CF = Conversion factor (10^{-6} kg/mg)
FI = Fraction ingested from contaminated source (site-specific)
EF = Exposure frequency (365 days/yr)
ED = Exposure duration (median, 9 yr; upper bound, 30 yr)
BW = Body weight (70 kg for adults; 16 kg for children 1-6 years old)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

The Guidelines do not give any default values for FI, but indicate that this value should be determined by contaminant location and activity patterns of the population.

The Exposure Factors Handbook (U.S. EPA, 1990d) gives the following equation for children who inadvertently ingest soil/dust from their hands or food while playing in or around their home. The range represents typical (50th percentile) and reasonable worst case (about 90th percentile) values for the expected distributions.

$$LADE = \frac{CR \times C \times ED}{BW \times LT \times (365\ days/yr)} \quad (2\text{-}21)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
CR = Ingestion rate: average, 0.2 g/day; range, 0.2-0.8 g/day
C = Chemical concentration in soil (mg/g)
ED = Exposure duration (average, 800 days; range, 800-2,200 days)
BW = Body weight (16 kg)
LT = Lifetime (75 yr)

Exposure duration was based on the estimate that contact with soil could occur from 75% of the time over a 3-yr period (typical) to 100% of the time over a 6-yr period (reasonable worst case).

For calculating Preliminary Remediation Goals for carcinogens and noncarcinogens (U.S. EPA, 1991c), an age-adjusted, time-weighted soil ingestion rate of 114 mg-yr/kg-day is used to estimate residential exposures over a 30-year period (350 days/yr). This estimate is based

on a 6-year exposure for children 1-6 years old (15 kg body weight) during which time the soil ingestion rate is estimated to be 200 mg/day, and a 24-year exposure for adults (70 kg body weight) during which time the soil ingestion rate is estimated to be 100 mg/day. This same approach has been proposed for calculating soil screening levels for carcinogens (U.S. EPA, 1994d); however, for calculating soil screening levels for noncarcinogens, EPA has proposed that this value be based on exposure to children age 6 and younger as this is the subpopulation at highest risk (U.S. EPA, 1994d). For this age group, the recommended default exposure parameters are: 15 kg body weight, 200 mg/day soil ingestion rate, 350 days/yr exposure frequency, and 6 years for exposure duration and averaging time.

2.5.2.6 Inhalation of vapors outside the home

The Exposure Factors Handbook (U.S. EPA, 1990d) provides the following equation for calculating exposure resulting from the inhalation of contaminated air outside the home.

$$LADE = \frac{IR \times C \times ED}{BW \times LT \times (365\ days/yr)} \qquad (2\text{-}22)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
IR = Inhalation rate (average, 1.4 m^3/hr; range, 1.4-3.0 m^3/hr)
C = Chemical concentration in air ($\mu g/m^3$)
ED = Exposure duration (average, 1,440 hours; range, 1,440-4,800 hours)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

The average exposure duration was based on an average of 3.07 hr/day spent outside the home for 52 weeks per year for 9 years. The reasonable worst case was based on an average of 3.07 hr/day spent outside the home for 52 weeks per year for 30 years. The inhalation rate was based on an average for a mixture of outdoor activities.

2.5.2.7 Inhalation of vapors inside the home

The Exposure Factors Handbook (U.S. EPA, 1990d) provides the following equation for estimating exposure resulting from the inhalation of contaminated air inside the home.

$$LADE = \frac{IR \times C \times ED}{BW \times LT \times (365\ days/yr)} \qquad (2\text{-}23)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
IR = Inhalation rate: average, 0.63 m^3/hr; range, 0.63-0.89 m^3/hr
C = Chemical concentration in air ($\mu g/m^3$)
ED = Exposure duration (average 54,000 hours; range, 54,000-180,000 hr)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

The average exposure duration is based on 115 hr/week spent inside the home, 52 weeks/yr for 9 years. The reasonable worst case is based on an average of 3.07 hr/day spent inside the home, 52 weeks/yr for 30 years. The inhalation rate was based on an average for a mixture of indoor activities.

Finley et al. (1993) present a modified version of this equation, using estimates of contaminant concentrations in the shower air, bathroom air and household air together with estimates of the time spent in the shower, bathroom and home to calculate a time-weighted average exposure.

2.5.2.8 Inhalation of vapors while showering

The Superfund Guidelines (U.S. EPA, 1989a) provide the following equation for estimating exposure resulting from inhalation of contaminated air while showering:

$$Intake = \frac{CA \times IR \times ET \times EF \times ED}{BW \times AT} \qquad (2\text{-}24)$$

where:

Intake = mg/kg-day
CA = Chemical concentration in air (mg/m^3)
IR = Inhalation rate
  20 m^3/day, average for adults
  30 m^3/day, upper bound for adults
  age-specific values (U.S. EPA, 1985a)
  0.6 m^3/hr, all age groups
ET = Exposure time (7 min/shower, 50th percentile; 12 min/shower, 90th percentile)
EF = Exposure frequency (pathway specific)
ED = Exposure duration (residential - median, 9 years; upper bound, 30 years; lifetime, 70 years)
BW = Body weight (70 kg for adults, age-specific body weights for other age groups)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

The Guidelines do not give a default value for EF.

The Exposure Factors Handbook (U.S. EPA, 1990d) gives a similar equation:

$$LADE = \frac{IR \times C \times ED}{BW \times LT \times (365\ days/yr)} \quad (2\text{-}25)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
IR = Inhalation rate: 0.6 m^3/hour
C = Chemical concentration in air ($\mu g/m^3$)
ED = Exposure duration (7 min/shower/day for 9 yr = 375 hr 9 average; range, 375-2,200 hr)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

The range for exposure duration represents typical and reasonable worst case values:

2.5.2.9 Inhalation of particulates outside the home

The Exposure Factors Handbook (U.S. EPA, 1990d) gives the following equation for estimating exposure resulting from inhalation of contaminated particulate matter outside the home:

$$LADE = \frac{IR \times PC \times RF \times C \times ED \times 10^{-6}\ g/\mu g}{BW \times LT \times (365\ days/yr)} \quad (2\text{-}26)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
IR = Inhalation rate (average, 1.4 m^3/hr; range, 1.4-3.0 m^3/hr)
PC = Particulate concentration in air ($\mu g/m^3$) (site specific)
RF = Respirable fraction of particulates (site specific)
C = Concentration of chemical on particulate ($\mu g/g$) (site specific)
ED = Exposure duration (average, 1,440 hr; range, 1,440-24,800 hr)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

The exposure duration was based on an average of 3.07 hr/day spent outside the home for 52 weeks per year for 9 years. The reasonable worst case was based on an average of 3.07 hr/day spent outside the home for 52 weeks per year for 30 years. The inhalation rate was based on an average for a mixture of outdoor activities.

2.5.2.10 Inhalation of particulates inside the home

The Exposure Factors Handbook (U.S. EPA, 1990d) gives the following equation for estimating exposure resulting from inhalation of contaminated particulate matter inside the home:

$$LADE = \frac{IR \times PC \times RF \times C \times ED \times 10^{-6}\ g/\mu g}{BW \times LT \times (365\ days/yr)} \qquad (2\text{-}27)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
IR = Inhalation rate (average, 0.63 m^3/hr; range, 0.63-0.89 m^3/hr
PC = Particulate concentration in air ($\mu g/m^3$) (site-specific)
C = Concentration of chemical on particulate ($\mu g/g$) (site specific)
RF = Respirable fraction of particulates (site-specific)
ED = Exposure duration (average, 54,000 hr; range, 54,000-180,000 hr)
BW = Body weight (70 kg)
LT = Lifetime (75 yr)

The average exposure duration was based on an average of 115 hr/week spent inside the home, 52 weeks/yr for 9 years. The reasonable worst case was based on an average of 3.07 hr/day spent inside the home, 52 weeks/yr for 30 years. The inhalation rate was based on an average for a mixture of indoor activities.

2.5.2.11 Dermal contact with water while swimming

The Superfund Guidelines (U.S. EPA, 1989a) provides the following equation for estimating exposure resulting from dermal absorption of contaminants while swimming.

$$AD = \frac{CW \times SA \times PC \times ET \times EF \times ED \times CF}{BW \times AT} \qquad (2\text{-}28)$$

where:

AD = Absorbed dose (mg/kg-day)
CW = Chemical concentration in water (mg/L)
SA = Skin surface area available for contact (cm^2) (age-specific; see Section 2.4.3.2)
PC = Dermal permeability constant (cm/hr), chemical-specific
ET = Exposure time (2.6 hr/day, national average for swimming)
EF = Exposure frequency (7 days/yr, national average)
ED = Exposure duration (9 yr, 50th percentile, 30 yr, 90th percentile)
CF = Volumetric conversion factor for water (1 Liter/1000 cm^3)
BW = Body weight (70 kg for adults, age-specific values for other age groups)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

The Guidelines recommend that the 95th or 90th percentile values for contact rate and exposure frequency and duration be used in the equation and the 50th percentile value for the skin surface area available for contact.

For estimating dermal exposures resulting from showering, Finley et al. (1993) include another variable, the fraction of skin in contact with water. From the studies of McKone and Bogen (1991), Finley et al. estimated that this value would range from 0.4 to 0.9.

2.5.2.12 Dermal contact with soil while gardening

The Superfund Guidelines (U.S. EPA, 1989a) provide the following equation for estimating exposure/dose resulting from dermal contact with soil:

$$AD = \frac{CS \times CF \times SA \times AF \times ABS \times EF \times ED}{BW \times AT} \quad (2\text{-}29)$$

where:

AD = Absorbed dose (mg/kg-day)
CS = Chemical concentration in soil (mg/g)
CF = Conversion factor, 10^{-6} kg/mg
SA = Skin surface area available for contact (cm^2/event)
AF = Soil-to-skin adherence factor (site-specific)
ABS = Absorption factor - chemical specific
EF = Exposure frequency (events/yr)
ED = Exposure duration (residential: 9 years, 50th percentile; 30 years, 90th percentile)
BW = Body weight (70 kg for adults, age-specific values for other age groups)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

The Guidelines indicate that the 50th percentile of the age-specific total surface area of the exposed body parts be used in the equation for SA (see section 2.4.3.11). The Guidelines do not give a default value for EF, but indicate that this value should be determined by local weather conditions and age of potentially exposed population. The soil-to-skin adherence factor (AF) is determined by the type of soil at a given site. EPA suggests that a value of 0.2 mg/cm^2 for AF is likely to be the best value to represent an average over all exposed skin and 1 mg/cm^2 may be a reasonable upper value (U.S. EPA, 1992a; 1992b). The Commonwealth of Massachusetts uses 0.51 mg/cm^2 as its default value for soil-to-skin adherence (MDEQE, 1989). Paustenbach et al. (1992) has also suggested that 0.5 mg/cm^2 is a reasonable estimate for this parameter. Thompson et al. (1992) used a uniform distribution with low and high values of 0.75 and 1.25 mg/cm^2 in applying a Monte Carlo simulation to dermal exposures.

A default value for the skin absorption factor (ABS) has not been adopted agency-wide by EPA; however, EPA Region IV has recommended using a value of 1.0% for organic compounds and 0.1% for inorganic compounds; these values take into account the soil matrix effect (U.S. EPA, 1992c). Based on the studies of Ryan et al. (1987), the state of Massachusetts recommends the following defaults for dermal absorption: 10-25% for VOCs; 1-10% for SVOCs; 1-10% for pesticides, and 0.1-1% for inorganics (MDEQE, 1989). In demonstrating the usefulness of Monte Carlo simulations in evaluating dermal exposures, Thompson et al. (1992) estimated skin absorption using the dermal uptake model of McKone (1990). McKone's model incorporates chemical properties, soil properties and skin properties to estimate absorption.

2.5.2.13 Inhalation of contaminants volatilized from water in the home

The following equation for estimating exposure resulting from inhalation of contaminants volatilized from water in the home was derived from EPA's risk equation for Preliminary Remediation Goals (U.S. EPA, 1991c):

$$LADE = \frac{C \times IR \times K \times EF \times ED}{BW \times AT \times (365\ days/yr)} \qquad (2\text{-}30)$$

where:

LADE = Lifetime Average Daily Exposure (mg/kg-day)
C = Chemical concentration in air (mg/L)
IR = Inhalation rate: 15 m^3/day, indoors
K = Volatilization factor (0.5 L/m^3; derived from Andelman, 1990)
EF = Exposure frequency, 350 days per year
ED = Exposure duration (30 years) residential occupancy
BW = Body weight (70 kg)
AT = Averaging time; lifetime (70 yr) for carcinogens, 30 years for noncarcinogens

2.5.2.14 Ingestion of chemicals in surface waters while swimming

The Superfund Guidelines (U.S. EPA, 1989a) provide the following equation for estimating exposure resulting from ingestion of contaminants in surface waters while swimming.

$$Intake = \frac{CW \times CR \times ET \times EF \times ED}{BW \times AT} \qquad (2\text{-}31)$$

where:

Intake = mg/kg-day
CW = Chemical concentration in water (mg/L)
CR = Contact rate (50 mL/hr)
ET = Exposure time (hr/event) (pathway-specific)
EF = Exposure frequency (7 days/yr, national average for swimming)
ED = Exposure duration (residential - 9 years, 50th percentile; 30 years, 90th percentile)
BW = Body weight (70 kg for adults, age-specific values for other age groups)
AT = Averaging time (ED x 365 days/yr for noncarcinogens, 70 yr for carcinogens)

The Guidelines do not give a default value for ET.

2.5.3 Exposure to Radionuclides

The Superfund Guidelines provide general information for assessing exposure to radioactive substances (U.S. EPA, 1989a, Chapter 10). More specific guidance can be found in "Background Information Document, Draft EIS for Proposed NESHAPS for Radionuclides" (U.S. EPA, 1989c).

Exposure assessments for radioactive substances differ fundamentally from those for nonradioactive substances in that it is not the mass of the substance deposited in a specific organ or tissue that causes the effect, but rather the total energy impacting that organ or tissue, regardless of whether the energy is from a radionuclide taken into the body (internal exposures) or from a radioactive source external to the organism (external exposures). Since the energy released is proportional to the decay rate of the radionuclide, quantities or concentrations of radionuclides are expressed not terms of mass, but rather in terms of units of activity (i.e., disintegrations per sec). The international standard for activity is the becquerel (Bq) which is equal to 1 disintegration per sec (and 2.7×10^{-11} Curies).

For internal exposures to radionuclides, intake is expressed in units of activity (becquerels) entering the organism, and the absorbed dose is expressed as the amount of energy imparted to a unit mass of tissue (in joules/kg = grays). Radionuclides emit various types of radiation such as alpha, beta or gamma radiation, each of which has a different level of effectiveness in causing a biological response. The concept "dose equivalent" takes into

consideration both the amount of energy deposited in a unit mass of a specific organ or tissue, as well as the biological effectiveness of the radiation (U.S. EPA, 1989a):

$$H = D \cdot Q \cdot N \qquad (2\text{-}32)$$

where:
H = dose equivalent [rems or sieverts: 1 sievert (Sv) = 100 rems]
D = absorbed dose [rads (1 rad = 100 ergs/g or 0.1 Gy) or grays, Gy (1 Gy = 1 joule/kg)]
Q = quality factor (i.e., 20 for alpha particles, 10 for neutrons and protons, and 1 for beta particles, positrons, x-rays, and gamma rays.
N = modifying factor (= 1)

Because radionuclides may remain in the body for an extended period of time, the radiation dose released from a radionuclide may continue long after the intake of the nuclide has ceased. Therefore, the internal dose to a specific organ or tissues is typically reported in terms of the "committed dose equivalent" ($H_{T,50}$) which is defined as the integral of the "dose equivalent" in a particular tissue T for 50 years after intake. A 50-year period is used to correspond to a working lifetime.

The "effective dose equivalent" is the weighted sum of the "dose equivalents" to all organs and tissues:

$$H_E = \sum_T w_T \cdot H_T \qquad (2\text{-}33)$$

where:
H_E = effective dose equivalent (rems or Sieverts)
w_T = weighting factor for organ or tissue T
H_T = mean dose equivalent for organ tissue T

The weighting factor corresponds to the fractional contribution of organ or tissue T to the total health risk. Standardized w_T values are 0.25 for gonads, 0.15 for breast, 0.12 for red marrow, 0.12 for lungs, 0.03 for thyroid, 0.03 for bone surface and 0.06 for each of the five remaining organs receiving the highest doses.

The "committed effective dose equivalent" is defined as the weighted sum of the "committed dose equivalents" to all organs and tissues:

$$H_{E,50} = \sum_T w_T \cdot H_{T,50} \quad (2\text{-}34)$$

where:
$H_{E,50}$ = committed effective dose equivalent (rems or Sieverts)
w_T = weighting factor for organ tissue T
$H_{T,50}$ = mean committed dose equivalent for organ tissue T

2.5.3.1 Internal Exposures

Inhalation and ingestion are important exposure pathways for internal exposure to radionuclides. In such cases, the "dose equivalent" to a specific organ can be estimated by multiplying the amount of each radionuclide inhaled or ingested times the appropriate dose conversion factor (DCF). The DCF is expressed as the "dose equivalent" per unit intake.

$$H = [C \cdot IR \cdot ED \cdot EF \cdot] \; x \; DCF \quad (2\text{-}35)$$

where:
H = dose equivalent (Sieverts)
C = concentration (e.g., Bq/L drinking water)
IR = intake rate (e.g., L/day for drinking water)
ED = exposure duration (e.g., years)
EF = exposure frequency (days/yr)
DCF = dose conversion factor (e.g. Sievert/Bq)

If the concentration of the radionuclide is measured in terms of mass (i.e., mg/L), it can be converted to becquerels by the use of the specific activity value (Bq/mg) for that nuclide.

A similar approach can also be used to derive the "effective dose equivalent" which, as noted above, is the weighted sum of the "dose equivalents" to all irradiated organs and tissues. DCFs for both inhalation and ingestion exposures to radionuclides have been developed for regulation of occupational exposures (U.S. EPA, 1988d; Eckerman and Ryman, 1993); however, EPA notes that these DCFs may not be appropriate for the general population (U.S. EPA, 1989a). Instead, EPA recommends modifying the standard exposure equations (see Section 3.5.2) by deleting the body weight and averaging time from the denominator, to obtain estimates of the intakes (in Bq) of the radionuclides of concern. In the Superfund methodology these estimates are compared with slope factors expressed in terms of Risk/Bq. The slope factors, which represent age-averaged lifetime excess cancer incidence per unit intake of radionuclide, are derived from a model that accounts for the amount of each radionuclide absorbed into the body through the gastrointestinal tract or through the lungs, the distribution and retention of each radionuclide in body tissues and organs, and the age, sex, and weight of the individual at the time of the exposure.

Depending on the characteristics of the radionuclides of concern, consideration of radioactive decay and ingrowth of decay products can be important factors in evaluating overall exposure and risk. Specific environmental fate and transport models have been developed for radionuclides which incorporate radioactive decay and ingrowth of decay products into predictions of dose and risk (see Till and Meyer, 1983; Miller, 1984, NCRP, 1984a; NCRP, 1989, U.S. EPA, 1989c). If decay products are present in secular equilibrium (i.e., equal activity concentrations) with their respective parent isotopes, then these isotopes must be included in the risk assessment.

2.5.3.2 External Exposures

External radiation doses involve exposures to radiation or high energy particles released from radionuclides without the radionuclide itself becoming incorporated into the organism. Gamma and x-rays are the most penetrating forms of radiation. External exposure to beta particles can impart a dose to the outer layer skin cells, although high energy beta radiation can penetrate the human body. Alpha particles are not sufficiently energetic to penetrate the outer layer of the skin.

Potential external exposure pathways to be considered include immersion in contaminated air or contaminated water, and radiation exposure from ground surfaces contaminated with beta- and photon-emitting radionuclides. External exposures may be determined by monitoring and sampling for radionuclides in environmental media, direct measurement of radiation fields, or by mathematical modeling (U.S. EPA, 1989a). Monitoring data can provide estimates of dose rates (i.e., Sieverts/hr) which when multiplied by duration of exposure gives a estimate of "dose equivalents". Alternatively, measured or predicted concentrations of radionuclides in environmental media (Bq/m^3 for air or water and Bq/m^2 for soil) can be multiplied by DCFs and exposure duration. The DCFs in these cases relate radionuclide concentrations on the ground, in air, or in water to external dose rates (e.g., Sv/sec per Bq/m^2 for ground contamination, or Sv/sec per Bq/m^3 for air or water immersion).

2.5.4 Multimedia and Multipathway Exposures

2.5.4.1 Baseline Risk Assessments

In baseline risk assessments conducted according to the Superfund guidelines, exposure/dose (intake) is calculated for individual pathways using the equations listed in Section 2.5.2. The total dose resulting from exposure across all relevant pathways is not calculated. Instead, the chronic daily intake (CDI) for each pathway is divided by the

appropriate reference toxicity value (RfD) to arrive at a Hazard Quotient for each noncarcinogenic contaminant (U.S. EPA, 1989a). The reference toxicity values used in the calculation are selected to correspond to the expected duration of the exposure (i.e., a chronic RfD is used for exposure durations of 7 years to a lifetime; a subchronic RfD is used for exposure durations of 2 weeks to 7 years, and 1 or 10-day Drinking Water Health Advisories are used for exposures less than 2 weeks). Hazard Quotients for exposure to several different contaminants by the same pathway are summed to arrive at a Hazard Index, and Hazard Indices for different pathways are summed to derive a "Total Exposure Hazard Index". The latter step is recommended only for exposure pathways that can be reasonably combined and only when the same individuals would consistently face the "reasonable maximum exposure" for each of the pathways. A Total Exposure Hazard Index is calculated separately for chronic, subchronic, and short-term exposures.

EPA's Office of Research and Development has stated that conversion from a concentration in air to an internal dose "is not always appropriate" (U.S. EPA, 1994d); consequently, exposures from inhalation pathways should be evaluated separately from other pathways.

2.5.4.2 Preliminary Remediation Goals for Non-Radionuclides

Part B of the Superfund Guidelines (U.S. EPA, 1991c), describe the procedures for deriving Preliminary Remediation Goals. In the PRG methodology, multimedia pathways are used to derive media concentrations at which the total exposure/dose would not result in a Hazard Index greater than 1 or a total carcinogenic risk greater than 10^{-6}. As part of this process, two cases of multimedia exposures are evaluated; (1) residential exposures resulting from ingestion of contaminated drinking water (derived from ground water or surface water) and inhalation of volatiles released from potable water inside the home; and (2) exposures at industrial/commercial sites resulting from ingestion of soil and inhalation of volatiles released from soil or inhalation of soil particles re-suspended in the air. The combined exposure/dose equation for the first case is as follows:

$$Intake = \frac{CW \times IR_w \times EF \times ED}{BW \times AT \times 365\ days/yr} + \frac{CW \times K \times IR_i \times EF \times ED}{BW \times AT \times 365\ days/yr} \qquad (2\text{-}36)$$

where:

Intake = mg/kg-day
CW = Chemical concentration in water (mg/L)
IR_w = Ingestion rate; for adults 2 L/day
EF = Exposure frequency (350 days/yr for residents)
ED = Exposure duration (30 years)
BW = Body weight (70 kg)
AT = Averaging time (30 yr for noncarcinogens)
K = Volatilization factor (0.5 L/m^3; derived from Andelman, 1990)
IR_i = Inhalation rate (15 m^3/day)

For occupational exposures resulting from ingestion of soil and inhalation of volatiles released from soil or contaminated soil particles re-suspended in the air, the combined exposure/dose equation presented in Part B of the Superfund Guidelines is as follows:

$$Intake = \frac{C \times IR_s \times 10^{-6}\ kg/mg \times EF \times ED}{BW \times AT \times 365\ days/yr} + \frac{C \times IR_a \times EF \times ED \times (1/VF + 1/PEF)}{BW \times AT \times 365\ days/yr} \quad (2\text{-}37)$$

where:

Intake = mg/kg-day
C = Chemical concentration in soil (mg/kg), site-specific
IR_s = Soil ingestion rate (50 g/day)
EF = Exposure frequency (250 days/yr)
ED = Exposure duration (25 yr)
BW = Body weight (70 kg)
AT = Averaging time (25 yr, always equal to ED for noncarcinogens)
IR_a = Inhalation rate (20 m^3/day)
VF = Soil-to-air volatilization factor (m^3/kg, chemical specific)
PEF = Particulate emission factor (m^3/kg, see below)

Methods for deriving VF and PEF have been recently revised (U.S. EPA, 1994d). The PRG methodology has been re-proposed as the approach to use to derive soil screening levels (U.S. EPA, 1994d); however, in response to the concerns from EPA's Office of Research and Development, the ingestion and inhalation pathways are evaluated separately (U.S. EPA, 1994d).

2.5.4.3 Preliminary Remediation Goals for Radionuclides

Part B, Chapter 4 of the Superfund Guidelines (U.S. EPA, 1991c), describes the procedures for deriving Preliminary Remediation Goals for radionuclides. The multipathway exposures evaluated for radionuclides include (1) residential exposures resulting from ingestion of contaminated drinking water (derived from ground water or surface water) and inhalation of volatiles released from potable water inside the home; (2) residential exposures resulting from direct ingestion of soil as well as external exposure to gamma radiation, and (3) exposures at industrial/commercial sites resulting from ingestion of soil and inhalation of volatiles radionuclides released from soil, inhalation of radioactive soil particles re-suspended in the air, and external exposure due to gamma emitting radionuclides. The combined radiation intake equation for the first case is as follows:

$$Intake = [RW \times IR_w \times EF \times ED] + [RW \times K \times IR_i \times EF \times ED] \quad (2\text{-}38)$$

where:

Intake = pCi
RW = Radionuclide concentration in water (pCi/L)
IR_w = Ingestion rate; for adults 2 L/day
EF = Exposure frequency (350 days/yr for residents)
ED = Exposure duration (30 years)
K = Volatilization factor (0.5 L/m^3; derived from Andelman, 1990)
IR_i = Inhalation rate (15 m^3/day)

For residential exposures resulting from direct ingestion of soil and external exposure to gamma radiation, the radiation dose equation can be expressed as:

$$Intake = [RS \times IF_s \times 10^{-3} g/mg \times EF] + [RS \times ED \times 10^{3} g/kg \times D \times SD \times (1 - S_e) \times T_e] \quad (2\text{-}39)$$

where:

Intake = pCi
RS = Radionuclide concentration in soil (pCI/g)
IF_s = Age-adjusted soil ingestion rate (3600 mg-yr/day)
EF = Exposure frequency (350 days/yr)
ED = Exposure duration (30 yr)
D = Depth of radionuclides in soil (0.1 m default)
SD = Soil density (default, 1.43 x 10^3 kg/m^3
S_e = Gamma shielding factor (default, 0.2)
T_e = Gamma exposure factor (default, 1)

For occupational exposures resulting from (1) ingestion of soil, (2) inhalation of volatiles released from soil, (3) volatiles released from re-suspended soil particles, and (4) external exposure due to gamma emitting radionuclides, the combined radiation dose equation is as follows:

$$\begin{aligned} Intake = & [RS \times ED \times EF \times IR_s \times 10^{-3}\ g/mg] \\ & + [RS \times ED \times EF \times IR_a \times 10^{3}\ g/kg \times (1/VF)] \\ & + [RS \times ED \times EF \times IR_a \times 10^{3}\ g/kg \times (1/PEF)] \\ & + [RS \times ED \times 10^{3}\ g/kg \times D \times DS \times (1-S_e) \times T_e] \end{aligned} \qquad (2\text{-}40)$$

where:

Intake = pCi
RS = Radionuclide concentration in soil (pCI/g)
IR_s = Soil ingestion rate (50 g/day)
EF = Exposure frequency (250 days/yr)
ED = Exposure duration (25 yr)
IR_a = Inhalation rate (20 m^3/day)
VF = Soil-to-air volatilization factor (m^3/kg, radionuclide specific)
PEF = Particulate emission factor (default, 4.63 x 10^9 m^3/kg, see below)
D = Depth of radionuclides in soil (0.1 m default)
SD = Soil density (default, 1.43 x 10^3 kg/m^3
S_e = Gamma shielding factor (default, 0.2)
T_e = Gamma exposure factor (default, 1)

2.5.5 Selecting Exposure/Dose Values for Use in Risk Assessments

Two types of exposure/dose estimates are recommended by EPA for use in risk assessments; bounding estimates and estimates of values within an actual exposure frequency distribution.

2.5.5.1 Bounding Estimates

EPA suggests that first step in an exposure assessment be the identification of bounding estimates for individual exposure pathways (U.S. EPA, 1992a). The method used for bounding estimates is to postulate a set of values for the parameters in the exposure or dose

equation that will result in an exposure or dose higher than any exposure or dose expected to occur in the actual population. If the value of this estimate is not significant, the exposure pathway can be eliminated from further consideration. One type of bounding estimate is the "theoretical upper bounding estimate" (TUBE) which is calculated by assuming limits for all variables used to calculate exposure or dose that, when combined, will result in the mathematically highest exposure or dose. Bounding estimates can eliminate pathways from further consideration but they cannot determine whether a pathway is significant.

Bounding estimates can be used as a screening step in a exposure assessment. Screening assessments may be site-specific, they may involve a comparison between several sites, or they may be generic to a certain type of site (i.e., an industrial segment or climatic region) (U.S. EPA, 1992a).

EPA has recently proposed generic soil screening levels for 105 chemicals based on this concept of bounding estimates (U.S. EPA, 1994d). These screening levels are derived from generic algorithms using conservative assumptions such that the predictions "overestimate potential conditions and thereby limit false negative results".

2.5.5.2 Exposure Estimates Based on Distributions of Parameter Values

Exposure distribution curves are cumulative probability distributions characterizing interindividual variability in exposures. EPA recommends that exposure estimates of central tendency and estimates of the upper end of the distribution curve be considered in exposure assessments. The central tendency (CT) may be either the arithmetic mean or the median (U.S. EPA, 1992a). The arithmetic mean can be approximated by using average values for all the appropriate exposure parameters, but it does not represent any particular point on the exposure distribution curve. The median estimate corresponds to the 50th percentile of the distribution. For highly skewed data arrays, the median value can be approximated by calculating the geometric mean.

High-end exposure estimates (HEEE), also called reasonable maximum exposures (RME) in the Superfund Guidelines, are plausible estimates of exposure for those individuals at the upper end of an actual exposure distribution. Conceptually, the HEEE would be above the 90th percentile of the population distribution, but not higher than the individual in the population who has the highest exposure. The combination of values assigned to the exposure or dose parameters should be expected to be found in the actual population, particularly in regard to future-use scenarios. The HEEE can be chosen in one of several ways:

- If sufficient data on the distribution of doses are available, the value can be taken directly from the percentiles of interest within the high end (i.e., >90th percentile).

- The value can be taken from an estimate based on a simulation of the distribution (derived from an exposure model or a Monte Carlo simulation) if sufficient data used to characterize the exposures are available.

- The value can be estimated by using maximum or near maximum values for one or more of the most sensitive variables, leaving the others at their mean values.

- The value can be estimated by starting with a bounding estimate and backing off the limits used until the combination of parameter values is, in the judgement of the assessor, clearly within the distribution.

In regard to the use of simulated distributions, EPA (1990d) notes that "unless a great deal is known about exposures or doses at the high end of the distribution, simulated distributions may not be able to differentiate between bounding estimates and high-end estimates". "Simulations often include low probability estimates at the upper end that are higher than those actually experienced in a given population, due to improbability of finding these exposures or doses in a specific population of limited size, or due to nonobvious correlations among parameters at the high ends of their ranges". EPA recommends using a value somewhat less than the highest Monte Carlo estimated value. When simulated distributions use unbounded default parameter distributions, such as lognormal distributions, there will not be a finite maximum exposure limit for the simulation, so the maximum value will vary with repeated simulations. EPA's Science Advisory Board (U.S. EPA, 1992d) has recommended that when unbounded lognormal distributions are used as a parameter default, the high-end exposure estimate should not exceed the 99.9th percentile. However, the 99.9th percentile should not automatically be considered a bounding estimate, because such bounding estimates are expected to vary depending on the size of the population (U.S. EPA, 1992a).

EPA Approach - According to the Exposure Assessment Guidelines (U.S. EPA, 1992a), exposures should be evaluated in terms of central tendency (CT) and a high-end estimate. The high-end exposure estimate (HEEE) is conceptually equivalent to the "reasonable maximum exposure" (RME) used in Superfund risk assessments (U.S. EPA, 1993a). The RME is defined as "the highest exposure that is reasonably expected to occur at a site, and in practice is estimated by combining 90-95th percentile values for some but not all exposure parameters" (U.S. EPA 1993a). Note: For Superfund risk assessments, the 95% upper confidence limit on the arithmetic mean of the contaminant concentration was originally recommended for use in estimating exposure. However, in response to the recommendations of EPA's Science Advisory Board, EPA now suggests that frequency distributions of

contaminant concentrations and exposure parameters be used, whenever possible, to fully evaluate the most probable distribution pattern for exposure and risk (U.S. EPA, 1992a).

2.5.5.3 Types of Distributions of Parameter Values

If exposure distributions are estimated using a Monte Carlo simulation, the outcome of the simulation will vary with the data set and type of distribution that is used for each of the exposure parameters. The distributions can be categorized as follows (Gephart et al., 1994):

- Normal - normal or bell-shaped curve described by a mean and a standard deviation above zero.
- Lognormal - lognormal curve with no values less than 0 and described by a mean and standard deviation greater than 0.
- Cumulative - irregular probability distribution described by a minimum, maximum, and up to 25 points described by a point value, and point probability (greater than 0 and less than 1).
- Histogram - defined histogram distribution described by a minimum, maximum, and up to 25 equal-length classes, with each class given a probability weight.
- Triangular - a triangular distribution described by three points; minimum, most likely, and maximum.
- Uniform - a uniform distribution described by a minimum and maximum.

The distribution that utilizes the maximum amount of available information is the one chosen as the best descriptor (Gephart et al., 1994). If information on a specific parameter includes the minimum, maximum, most likely (mode), mean, and the data can be sorted into a frequency distribution, then a cumulative or histogram distribution would be the preferred choice. Normal or log normal distributions are used if there is confidence that the shapes of the curves are appropriate. Triangular and uniform distributions are only used when a minimal amount of information is available.

2.5.6 Estimates of Population Risk

The estimates of exposure/dose can be used to develop population frequency distributions from which descriptors of population risk can be derived. The EPA Exposure

Assessment Guidelines (U.S. EPA, 1992a) present two types of population risk descriptors; one based on the incidence of health effects and the second based on the fraction of the population above a specific level of risk.

2.5.6.1 Incidence of health effects

This risk descriptor is "the probabilistic number of health effects cases estimated in the population of interest over a specified time period" (U.S. EPA, 1992a). For noncarcinogens and nonlinear threshold carcinogens, it can be obtained by summing all the individual risks over the entire population. For linear nonthreshold carcinogens, it can be obtained by multiplying the slope factor, the arithmetic mean of the dose, and the population size. The latter method can be used only if the risk model assumes a linear nonthreshold response and individual risk is not higher than about 10^{-1}.

2.5.6.2 Fraction of the population above a specific risk level

This risk descriptor is "an estimate of the percentage of the population, or the number of persons, above a specified level of risk, RfD, RfC, LOAEL, or other specific level of interest" (U.S. EPA, 1992a). This estimate can be obtained by measuring or simulating the population distribution by: (1) measuring the distribution of the exposure or dose (if the population is small); (2) simulating the distribution using an exposure model or a Monte Carlo model; or (3) identifying and enumerating certain population segments known to be at a higher level of exposure, dose, sensitivity, or risk than the level of interest.

2.5.7 Uncertainty Analysis

Uncertainty in exposure assessments can be classified into three broad categories: scenario, parameter and model uncertainty (U.S. EPA, 1992a). Together these factors contribute to the overall uncertainty in the exposure estimate.

2.5.7.1 Scenario uncertainty

Scenario uncertainty refers to uncertainty associated with missing or incomplete data used to define exposure and dose (U.S. EPA, 1992a). The sources of scenario uncertainty include descriptive errors, aggregation errors, errors in professional judgement, and incomplete analysis. Descriptive errors refer to inaccurate information about sources of contamination. Aggregation errors can result from lumping together assumptions such as those concerning homogeneous populations and steady state conditions. Examples of possible errors in professional judgement are the selection of inappropriate exposure scenarios or environmental

fate models. Lack of analysis refers to incomplete assessment of all pertinent exposure pathways.

2.5.7.2 Parameter uncertainty

Parameter uncertainty relates to the degree of confidence, or lack therein, an assessor has in the parameter value used in calculating exposures/doses. Sources of parameter uncertainty include "measurement errors, sampling errors, variability, and use of generic or surrogate data" (U.S. EPA, 1992a). Measurement errors can be random or systematic; random errors result from imprecision in the measurement process; systematic errors reflect a bias or tendency away from the true value. Temporal or spatial variability in exposure parameters may be difficult to characterize adequately. Generic and surrogate data must be used if site-specific data are not available, and whether these accurately reflect the actual conditions can be a major source of uncertainty.

2.5.7.3 Model uncertainty

Model uncertainty, which is uncertainty regarding gaps in scientific theory required to make predictions on the basis of causal inferences, can result from relationship errors or modeling errors (U.S. EPA, 1992a). Relationship errors include errors in correlations between chemical properties, structure-reactivity correlations, and environmental fate models. Modeling errors are due to models being simplified representations of reality. Many existing models and the hypotheses contained within them cannot be fully tested, although certain components may be tested. Even when a model is validated under a certain set of conditions, uncertainty will exist in its application to situations beyond the test system. The uncertainties associated with the use of models can be assessed in several ways. Different models can be tested and the range of outputs can be considered to be representative of the uncertainty range. When the data base is adequate, the uncertainties can be assessed in terms of the level of model validation and verification. Validation is the process of examining the performance of the model compared to actual observations under situations representative of those being assessed (U.S. EPA, 1985c). Verification is the process of confirming that the model computer code is producing the proper numeric output.

2.5.7.4 Uncertainty in exposure estimates

EPA notes that in deriving estimates of exposure, the distinction must be made between variability in a population versus uncertainty in the estimate (U.S. EPA, 1992a). Variability in a population is the receipt of different levels of exposure by different individuals, whereas uncertainty in the estimate is due to lack of knowledge about the correct value for

a specific exposure measure or estimate. Estimates of exposure based on the "worst case scenario" do not attempt to distinguish between variability in the population and uncertainty in the estimate. By using high-end individuals (variability) and upper confidence bounds on data and physical parameters (uncertainty) the exposure estimates might be interpreted as "not exceeding an upper bound on exposure received by certain high-end individuals". However, EPA states that high-end estimates should be presented with both the upper and lower uncertainty bounds to better characterize the potential risks to the population (U.S. EPA, 1992a).

There are several approaches to characterizing uncertainty in exposure assessments (U.S. EPA, 1992a). One approach is an order-of-magnitude bounding of the parameter range when the uncertainty is high. Another involves a description of the range for each of the parameters including lower and upper bound and best estimates values. In some cases, the characterization of uncertainty can take the form of a probabilistic description of the parameter range. The appropriate characterization depends on whether there is evidence (i.e., from sensitivity analysis) that the results can be significantly affected by the variations within the parameter range. If this is the case, then an attempt should be made to reduce the uncertainty by developing a description of the likely occurrence of the values with the range which can be done by either statistical analysis if enough data points are available, or by expert judgement to develop a subjective probabilistic representation if only limited data points are available.

Most approaches for analyzing uncertainty have focused on techniques that examine how uncertainty in parameter values affect overall uncertainty in the assessment (U.S. EPA, 1992a). Several published reports (Cox and Baybutt, 1981, U.S. EPA, 1985b; Inman and Helton, 1988; Seller, 1987; Rish and Marnicio, 1988) have reviewed the methodology for evaluating uncertainty including sensitivity analysis, uncertainty propagation, probabilistic uncertainty analysis, and classical statistical methods.

Sensitivity analysis - Sensitivity analysis is the process of changing one variable (usually to its credible lower and upper bound estimates) while leaving the others constant (at their median value) and determining the effect on the output (U.S. EPA, 1992a). This method, which identifies those variables that have the greatest effect on exposure, can be used at the screening level to help focus further information gathering.

Analytical uncertainty propagation - This method involves examining how uncertainty in individual parameters affects the overall uncertainty of the exposure assessment. It requires an analysis of variance of individual parameters, explicit expressions of exposure, and the ability to either analytically or numerically obtain a mathematical derivative of the exposure/dose equation.

Probabilistic analyses - This method is exemplified by the Monte Carlo simulation which uses random numbers to select values from each parameter distribution (based on their probable occurrence in that distribution). The function is solved, and then the process is repeated numerous times with other sets of randomly generated values. The result is a distribution, or probability density function, for population exposure. The results of this analysis do not indicate which variables are the most important contributors to output uncertainty (U.S. EPA, 1992a). Latin Hypercube simulation is another approach for randomly sampling expected distributions of exposure parameters to generate a distribution of population exposure (AIHC, 1994). In this simulation, the random sampling is done within equal intervals of the distribution which minimizes the number of samples necessary to sufficiently represent the population. Latin Hypercube sampling maintains complete independence of the variables, and if any correlation exists between two variables, it must be specifically identified in the distribution descriptions (AIHC, 1994).

Classical statistical methods - This method can be used to analyze uncertainty in measured exposures (U.S. EPA, 1992a). The distribution can be measured directly or can be used to compute confidence interval estimates for percentiles of the exposure distribution. The latter are used to characterize uncertainty.

3. SAB AND NRC EVALUATIONS OF EPA GUIDELINES

3.1 SCIENCE ADVISORY BOARD (SAB)

At the request of the EPA's Office of Solid Waste and Emergency Response, the EPA Science Advisory Board (SAB) reviewed selected issues relevant to the Superfund risk assessment guidelines (U.S. EPA, 1993j). Several of the conclusions reached by SAB concerning exposure assessment methodology are summarized below.

- The SAB states that conceptual and practical difficulties exist in using a mean or the upper confidence limit (UCL) of the mean contaminant concentration in soil to derive RMEs (reasonable maximum exposures). These difficulties include the fact that (1) the UCL does not lead logically to a reasonable maximum exposure, particularly if a "hot spot" on a site was more likely to be visited by the potentially exposed population; (2) the UCL is combined with point estimates of other exposure parameters (50, 90 or

95th percentiles which makes the RME difficult to interpret; (3) The UCL does not take into account the spatial distribution of the contaminant; and (4) the calculation of a UCL requires statistical assumptions that are not generally met by sampling plans at a Superfund site.

- SAB states that the use of the mean contaminant concentration is appropriate for evaluating population risk (but not individual risk), but only if it is assumed that spatial and temporal variability in the concentration is small and that potential exposures are equally likely across the entire site. If these conditions are not met, SAB suggests looking at the distribution of average concentrations to which visitors at a site are exposed and taking a given percentile, such as the 90th percentile, of that distribution, as the concentration to be used in calculating the RME.

- If a mean contaminant concentration is used by EPA to evaluate exposure, SAB recommends using the arithmetic mean rather than the geometric mean.

- For evaluating soil contamination, the SAB recommends that EPA consider adopting statistical approaches which take into account the spatial distribution of the concentrations. Two such techniques are kriging and triangulation.

- The SAB recommends that EPA move towards a "full distributional approach" in calculating the RME. In this approach a distribution is developed for each input term in the exposure equation, and the overall distribution of exposures is calculated by using Monte Carlo methods. A particular percentile of this distribution, such as the 90th percentile, could then be used as the definition of the RME.

- The SAB recommends that EPA develop default distributions for those exposure parameters that are unlikely to vary significantly from site to site.

- Until new methods are adopted for calculating the RME, the SAB recommends that some type of "most reasonable" exposure also be calculated and be made available to risk managers.

- For the evaluation of exposures resulting from ingestion of soil, SAB supports the use of a 30-yr time-weighted average for the soil ingestion rate.

3.2 NATIONAL RESEARCH COUNCIL (NRC)

The National Research Council's report, "Science and Judgement in Risk Assessment" (NRC, 1994) is an evaluation EPA's risk assessment methodology. Although the report focuses specifically on hazardous air pollutants; many of the suggestions and recommendations contained in it, as they pertain to exposure assessments, can also be applied to risk assessments for other types of environmental contaminants.

NRC (1994) notes that EPA's revised exposure assessment guidelines (U.S. EPA, 1992a) have incorporated many of the concepts and recommendations presented in earlier NRC documents (NRC, 1983 and 1991), including, most importantly, the need to evaluate the distribution of individual and population exposures and to incorporate uncertainty analysis into the exposure assessment. In its most recent guidelines, EPA has replaced its traditional use of the "maximally exposed individual" (MEI) with two new descriptors of exposure, the "high-end exposure estimate" (HEEE) and the "theoretical upper bounding estimate" (TUBE). The TUBE is an estimate of exposure that is expected to exceed the levels of exposure experienced by all individuals in the actual distribution. NRC (1994) recommends that bounding estimates such as the TUBE only be used in screening assessments to determine if further analysis is necessary.

The MEI represented a theoretical upper bound on exposure and was calculated using a number of very conservative exposure assumptions (i.e., for air pollutants a 70-yr, 24 hr/day exposure with no allowance for indoor attenuation). In contrast, the HEEE is a plausible estimate of exposure for those individuals at the upper end of an actual exposure distribution curve. Conceptually, the HEEE is above the 90th percentile of the population distribution, but not higher than the individual in the population with the highest exposure. The HEEE is based on the estimation of the distribution of exposures that people might actually encounter. NRC points out that the exact percentile of the exposure distribution that should be used for the HEEE is not specified by EPA (the guidelines state that it should be chosen to be consistent with the population size in a specific application). NRC (1994, p. 10-16) recommends that this estimate of exposure be based on the 100[(N-1)/N]th percentile of the cumulative probability distribution characterizing interindividual variability, where N is the number of persons used to construct the exposure distribution. Of the methods that EPA has identified for calculating HEEE (see Section 2.5.4.2), NRC recommends that those based on actual exposure distributions or distribution simulations should be preferred over those based on point estimates. When point estimates are used, NRC suggests that EPA provide a rationale for how such point estimates are generated, offer evidence that such values yield reasonably consistent representations of the desired percentiles, and justify the choice of a percentile if it differs from that which corresponds to the expected value of exposure to the person most exposed to emissions (NRC, 1994).

According to NRC (1994), time-activity patterns and estimates of microenvironmental concentrations (particularly for air quality studies) should be considered in developing population exposure (and risk) distributions, particularly for long-term exposure scenarios. Both factors can be either measured or modeled. Although EPA has recently incorporated distributions of typical (50th percentile), and reasonable worst case (90th percentile) residence times (i.e., 9 and 30 years, respectively) into its exposure assessment, NRC recommends that in evaluating exposures relating to air pollutants the default residency assumption for the calculation of the maximally exposed person remain at the "mean of the current U.S. life expectancy, in the absence of supporting evidence otherwise". Other NRC recommendations are listed below:

- NRC states that EPA should use an tiered approach to risk assessment starting with "relatively inexpensive screening techniques - such as simple, conservative transport models - and then for chemicals suspected of exceeding de minimis risk move on to more resource-intensive levels of data gathering, model construction, and model application ... screening techniques must err on the side of caution". For further analysis, the committee supports EPA's development of distributions of exposure based on actual measurements, results from modeling, or both".

- "EPA should continue to regard the use of default options as a reasonable way to deal with uncertainty about underlying mechanisms in selecting methods and models for use in risk assessment". "Clarify defaults and the rationales for them, including defaults now 'hidden', and develop criteria for selecting and departing from defaults".

- EPA should cooperate with other agencies to choose and validate a common set of uncertainty distributions for the default assumptions it uses.

- "Despite the advantages of developing consistent risk assessments between agencies by using common assumptions...EPA should include other methods, if any, that might be more accurate".

- "EPA should use a specific conservative mathematical technique to estimate the highest exposure likely to be encountered by an individual in the exposure group of interest".

- EPA should "conduct pediatric risk assessments whenever children might be at greater risk than adults".

- Evaluate all routes of exposure to address multimedia issues". "EPA should take advantage of new developments and approaches to the analysis of multimedia transport and fate data".

- "EPA should more rigorously establish the predictive accuracy and uncertainty of its methods and models and the quality of data used in risk assessments".

- "EPA should conduct formal uncertainty analyses, which can show where additional research might resolve major uncertainties and where it might not".

- "EPA should develop guidelines for quantifying and communicating uncertainty (e.g., for models and data sets) as it occurs in each step in the risk-assessment process".

4. OTHER EXPOSURE ASSESSMENT APPROACHES

The following sections present information exposure assessment approaches currently used or recommended by various governmental and non-governmental groups. With minor exceptions, the general EPA methodology, including the exposure/dose equations listed in Section 2.5.2, are used by most groups. Differences exist primarily in the choice of values to use for specific input parameters as well as in the selection of the most appropriate endpoint to characterize population exposure. However, in several cases a modified approach has been proposed in which environmental partitioning models are used to derive transfer coefficients between environmental media (i.e., between soil and plants, plants and food animals, water and fish, etc). These transfer coefficients are then incorporated into the exposure/dose equations.

4.1 GOVERNMENTAL AGENCIES

4.1.1 Resource Conservation and Recovery Act - RCRA (U.S. EPA, 1989e)

Guidelines for assessing exposure at RCRA sites (U.S. EPA, 1989e) are generally similar to those used at CERLA (Superfund) sites.

In the guidelines for RCRA facility investigations (U.S. EPA, 1989e), the following pathways are identified as possible sources of exposure: ingestion and dermal contact with surface soils, ingestion of drinking water derived from groundwater or surface water, inhalation of contaminated air outside the home or inside the home as a result of volatilization from water, inhalation of subsurface gases leaking into residential basements, and ingestion of contaminated biota. The default exposure parameters used by RCRA are: for adults, 70 kg body weight, 70 year exposure period, 2 L/day drinking water ingestion rate, 20 m^3/day inhalation rate, and 100 mg/day soil ingestion rate; and for children, 16 kg body weight for a 5-year exposure period (ages 1-6) (U.S. EPA, 1989e).

The RCRA guidelines state that when a criterion is below detection limits, "the detection limit will be used as a default value when making comparisons to investigation data, unless acceptably determined modeling values can be applied (i.e., values from air dispersion models)" U.S. EPA, 1989e).

It should be noted that sites designated for cleanup under CERCLA, including federal reservations, may be subject to state enforcement actions brought under RCRA (e.g., federal courts have ruled that the state of Colorado has authority to set standards, exclusive of CERCLA, for Rocky Mountain Arsenal). Therefore, risk assessments conducted for such state-regulated sites, would have to follow the guidelines for exposure assessments adopted by that state.

4.1.2 Agency for Toxic Substances and Disease Registry (ATSDR, 1990)

The Agency for Toxic Substances and Disease Registry has developed draft guidelines for conducting health assessments (ATSDR, 1990). The recommended methodology is similar to that of EPA, including the use of point estimates for the relevant exposure parameters. The standard default values for body weight, lifespan and residential occupancy at one location are the same as those used for Superfund risk assessments. Exposure scenarios considered by ATSDR include the following: inhalation, water ingestion, soil ingestion, food ingestion, and dermal exposure to water and soil. Default values for selected exposure parameters as used by ATSDR are listed under the appropriate subsections of Section 2.4.3. The exposure/dose equations for the exposure pathways are similar to those used by EPA with the exception that the equations for inhalation and ingestion of soil or water (but not the one for food ingestion) include, in the numerator, the additional parameter, "absorption factor". When information for this parameter is not available, the recommended default value is 100%. In addition, the equation for contaminant uptake through dermal exposures to water includes in the numerator the permeability constant (a default value is not given), and the equation for dermal exposure to soil includes in the numerator the Bioavailability Factor, the default for which is 100%.

4.1.3 U.S. Department of the Army (Dacre et al., 1980; Small, 1988; Small, 1989)

A 1991 Army directive (DA, 1991) states that in conducting health risk assessments as part of the Installation Restoration Program or at Formerly Used Defense sites, the current EPA guidelines should be followed, including those in the Superfund Manual (U.S. EPA, 1989a) and also those in the RCRA Facility Investigation (RFI) Guidance (U.S. EPA, 1989e).

Under the sponsorship of the U.S. Army, a methodology was developed for deriving preliminary pollutant limit values (PPLV) from exposure data and environmental partitioning coefficients (Dacre et al., 1980). PPLVs are contaminant concentrations in environmental media that can be used as nonregulatory control limits. The methodology was revised by Small (1988) and incorporated into a computer program known as the Pollution Hazard Assessment System (Small, 1989).

The PPLV methodology treats each possible exposure pathway as a series of compartments. The assumption is made that successive compartments are at equilibrium, and that maximum concentration values and pseudo-steady state conditions exist within each medium. The equilibria can be expressed in terms of partition coefficients; i.e., the ratio of the contaminant concentration between pairs of successive compartments. The PPLV designed for evaluating exposure through ingestion can be expressed by the following equation:

$$D_T = \frac{f \times DFI \times C_a}{BW} \qquad (4\text{-}1)$$

where:

D_T = Acceptable daily dose (mg/kg/day)
f = Fraction of the total diet for specific food type
DFI = Daily food intake (kg/day)
C_a = Limiting chemical concentration (mg/kg) in food animal (i.e., cattle)
BW = Body weight (kg)

The contaminant concentration in the food animal (C_a) can be expressed in terms of the concentration in the plants (C_p) that the animals consume and the equilibrium partition coefficient for plants and animal ($C_a = K_{pa} \times C_p$). Similarly, C_p can be derived from the concentration of the contaminant in soil water (C_w) and the equilibrium partition coefficient between soil water and plants ($C_p = K_{wp} \times C_w$). Finally, C_w can be derived from the concentration of the contaminant in the soil (C_s) and the equilibrium partition coefficient between soil and soil water ($C_w = K_{sw} \times C_s$). Equation 4-1 can therefore be expressed in terms of the concentration in soil and the partition coefficients:

$$D_T = \frac{f \times DFI \times C_s \times K_{pa} \times K_{wp} \times K_{sw}}{BW} \qquad (4\text{-}2)$$

where:

D_T = Acceptable daily dose (mg/kg/day)
f = Fraction of the total diet for specific food type
DFI = Daily food intake (kg/day)
Cs_a = Limiting chemical concentration in soil (mg/kg)
BW = Body weight (kg)
K_{sw} = Partition coefficient for soil and soil-water
K_{wp} = Partition coefficient for soil-water and plants
K_{pa} = Partition coefficient for plants and animals

Solving the equation for C_s using a reference toxicity value for D_T results in the derivation of a PPLV for that specific pathway. Other pathways can be evaluated using the same methodology. The following pathways are currently included in the Pollution Hazard Assessment System:

- ingestion of water
- ingestion of fish
- ingestion of plants grown on soil irrigated with contaminated water
- ingestion of food animals raised on pasture irrigated with contaminated water
- ingestion of groundwater contaminated by migration of contaminants through soil
- ingestion of milk from cows drinking contaminated water
- ingestion of milk from cows pastured on contaminated soil
- ingestion of plants grown on contaminated soil
- ingestion of food animals consuming water leached from contaminated soil
- ingestion of water or fish from water bodies in which sediments are contaminated.
- ingestion, inhalation, and dermal contact with dusts (residential and occupational)
- inhalation of vapors released from soils (open and confined areas)
- dermal absorption of contaminants in water

Partition coefficients for the various pairs of compartments may be difficult to determine and they may also vary with site-specific conditions such as soil type (Dacre et al.,1980); however, methods for estimating partition coefficients are incorporated into the Pollution Hazard Assessment System (Small, 1989).

Default values used in the calculation of PPLVs are shown in Table 4-1.

Table 4-1. Default Parameters used in Calculating PPLVs	
Parameter	**Value**
Body weight	70 kg, adult; 12 kg, 2-yr-old child
Body surface area	1.8 m^2, adult; 0.72 m^2, child
Inhalation rate - house basement	1.5 m^3/day, adult; 0.75 m^3/day, child
Inhalation rate - outside, occupational	17 m^3/day
Water intake	1.6 L/day, adult; 1.0 L/day, child
Soil ingestion - normal conditions	74 mg/day, adult; 820 mg/day, child
Soil ingestion - dusty conditions, occupational	180 mg/day
Vegetable intake	17 g/day, adult; 8 g/day, child
Beef intake	100 g/day, adult; 40 g/day, child
Dairy intake	0.3 L/day, adult; 0.56 L/day, child
Fish intake	6.5 g/day, adult; 2.2 g/day, child

Source: Small, 1989

4.1.4 State of California (CAPCOA, 1987; Sedman, 1989; CAPCOA, 1993)

The California Air Pollution Control Officers Association (CAPCOA) has released a draft source assessment manual for toxic air pollutants (CAPCOA, 1987). The manual recommends the use of models for estimating exposure/dose through inhalation, ingestion, and dermal contact resulting from the releases of contaminants in stack emissions. An example of an appropriate model is the California Energy Commission Environmental Pathways Uncertainty and Screening (EPUS) model. EPUS incorporates emission data, environmental fate calculations and exposure data, and therefore requires site- chemical- and population-specific inputs, including: stack release rates, atmospheric dispersion factors, effective settling velocity, soil characteristics, rainfall rates, surface water factors, plant and animal factors, removal rate constants, environmental media transfer coefficients, intake rates, and gut partition factors. The gut partition factor is that fraction of the ingested contaminant that would be absorbed and is therefor equivalent to a Bioavailability Factor used by other agencies. If no information is available on the gut partition factor, the default assumption is 1. The manual notes that caution is needed in correctly determining the values (or distributions) for the input variables.

In 1993 CAPCOA issued revised risk assessment guidelines for its Air Toxics "Hot Spots" Program. Guidelines outline procedures for conducting risk assessments for air emissions as well as landfill sites. A screening assessment can be used as long as the assessment does not underestimate health risks. Besides the inhalation pathway, the primary non-inhalation pathways that are considered are: dermal exposure, water ingestion, crop ingestion following direct deposition, and soil ingestion. Indirect pathways to be considered include: mother's milk, fish ingestion, crop ingestion due to root uptake, and ingestion of meat, eggs and dairy products. The recommended default values for these calculations are shown in Table 4-2.

Table 4-2. Default Parameters Used by CAPCOA

Parameter	Value
Body weight	70 kg, adult
Inhalation rate	10 m^3/day
Water intake	2 L/day
Surface of exposed skin	4,656 cm^2
Soil loading	0.5 mg/cm^2
Lifetime soil ingestion	110 mg/day
Consumption of root crop	0.05 kg/day
Consumption of vine crop	0.25 kg/day
Meat intake	0.300 kg/day
Milk intake	0.100 kg/day
Fish intake	0.024 kg/day

Source: CAPCOA, 1993

In addition to the default values for the population-specific parameters listed in Table 3-2, CAPCOA (1993) lists chemical-specific values for dermal absorption, gastrointestinal absorption, soil half-life, and bioavailability for 27 contaminants (or classes of contaminants). Dermal absorption factors range from 0.1% for inorganics to 10% for organics. Gastrointestinal absorption factors are set at 100% for all contaminants, and a bioavailability factor was available for only one contaminant class (dioxin and furans).

In developing a soil contact exposure scenario for calculating Applied Action Levels (AAL) for the State of California's Site Mitigation Decision Tree Manual, Sedman (1989)

evaluated the available literature, and derived time-weighted average values for soil ingestion and dermal contact with soil. These values, 0.15 g/day soil ingested and 0.45 g/day for dermal contact with soil, were

based on estimates of age-specific changes in ingestion and contact rates, and on a 70-year lifetime exposure. The values were used in the following exposure/dose equation:

$$AAL = \frac{MEL}{(TF1 \times 0.15 g/day + TF2 \times 0.45 g/day)} \quad (4\text{-}3)$$

where:

MEL = Maximum Exposure Level
TFI = Toxicokinetic factor for ingestion
TF2 = Toxicokinetic factor for dermal contact

The toxicokinetic factors take into account differences in absorption rates through the two exposure pathways.

4.1.5 State of Massachusetts (MDEQE, 1989)

The state of Massachusetts has developed guidance for risk characterization at waste disposal sites (MDEQE, 1989). The guidelines address the following possible exposure pathways: inhalation of vapors and particulate matter; ingestion and dermal contact with soil; ingestion and dermal contact with drinking water; inhalation of volatiles released from drinking water; dermal contact with surface water; incidental ingestion of surface water; inhalation of volatiles released from surface waters; ingestion of food; and ingestion of mother's milk. Default values for specific exposure parameters are listed under the appropriate subsections of Section 2.4.3. The exposure/dose equations are basically the same as those used by EPA with the exception that in each one there is an added parameter in the numerator, the Bioavailability Adjustment Factor (BAF) (or Relative Absorption Factor, as used by EPA Region I). The BAF is the ratio of the absorption efficiency for the route and medium of exposure (e.g., dermal contact with soil) and the absorption efficiency for the route and medium of exposure for the experimental study which is the basis of the Reference Dose or the Potency Value for the chemical in question (e.g., ingestion of water). The BAF takes into account the differences between the actual absorption of the "free" contaminant and the ability of the contaminant to become "free" from the medium or matrix to which it might be bound.

4.2 NONGOVERNMENTAL GROUPS

4.2.1 Finley et al. (1993, 1994)

Finley et al. (1993, 1994) have suggested that point estimates of exposure parameters be replaced with "standard" distributions so that they can more easily be used in the Monte Carlo method of probabilistic modeling. They state that this would allow for the characterization of the uncertainty and variation of the resulting exposure estimates. They also suggest performing a sensitivity analysis to determine which probability density functions have the greatest influence on the risk estimates so that the accuracy of the assessment can be qualitatively evaluated. Finley et al. (1993) evaluated cleanup goals for volatile organics in groundwater that might be used as a source of drinking water. The exposure pathways that they considered were: direct ingestion of tapwater, dermal contact while showering, inhalation of emissions from indoor fixtures and appliances, and ingestion of garden vegetables irrigated with tapwater. The distribution functions for exposure parameters used to estimate exposure through these pathways are shown in Table 4-3.

In their later report, Finley et al. (1994) note that for some exposure factors, such as tapwater consumption and body weight, parametric distributions (i.e., normal or lognormal), may "result in implausible values for even extreme individuals" when used in a Monte Carlo analysis. This is due to the fact that although the exposure parameters may have natural limits, the probability density function are unlimited. Finley et al. (1994) propose using non-parametric distributions, such as cumulative percentiles for some parameters unless the parametric distribution functions have been demonstrated to fit the original data using well-defined techniques. For certain parameters where the distribution data is especially limited they suggest using a triangular distribution (minimum, maximum, and mean) which can be viewed as "conservative characterizations of truncated normal or lognormal distributions". Finley et al. (1994) present recommended "standard" distributions for body weight, skin surface area, chronic inhalation rate, tapwater ingestion rate, fish consumption, soil ingestion, soil adherence to skin, residential occupancy period, time spent on the job, and shower duration.

4.2.2 Gephart et al., 1994

Gephart et al. (1994) evaluated the scientific documentation and distribution data available for various exposure parameters. The distributions were identified as being normal, lognormal, cumulative, histogram, triangular, or uniform. Gephart et al. (1994) recommend specific types of distributions and point values as shown in Table 4-4.

Table 4-3. Distribution Functions Used by Finley et al. (1993)						
Parameter	Distribution	Mean	Standard deviation	Min.	Max.	Ref.
INGESTION:						
Tap water ingestion (L/day)	empirical	1.1	-	0.5	5.5	b
Vegetable ingestion (kg/day)	lognormal	0.062[a]	1.8	-	-	c
Fraction homegrown	empirical	-	-	18%	47%	f
DERMAL CONTACT:						
Skin surface (cm^2)	correlation	-	-	14,200	21,500	-
Shower exposure time (hr/day)	lognormal	0.13[a]	0.085	-	-	d,e
Fraction of exposed skin (unitless)	uniform	-	-	0.4	0.9	e
INHALATION:						
Water use rate (L/hr):						
shower	lognormal	480[a]	160	-	-	d,e
house	lognormal	40[a]	15	-	-	d,e
Air exchange rate (m^3/hr)						
shower	uniform	-	-	4	20	d,e
bathroom	uniform	-	-	10	100	d,e
house	uniform	-	-	300	1200	d,e
Exposure time (hr/day):						
shower	lognormal	0.13[a]	0.085	-	-	d,e
bathroom	lognormal	0.33[a]	0.22	-	-	d,e
house	uniform	-	-	8	20	d,e
Inhalation rate (m^3/hr)	uniform	-	-	0.21	0.74	f

[a] Geometric mean
[b] Ershow and Cantor, 1989
[c] Salhotra et al., 1991
[d] James and Knuiman, 1987
[e] McKone and Bogen, 1991
[f] U.S. EPA, 1990d

Table 4-4. Default Values for Exposure Parameters Recommended by Gephart et al. (1994)			
Parameter	**Point value**	**Recommended Distribution**	**Data quality**
Adult body weight	72 kg	Cumulative	High
Child body weight	13 kg	Cumulative	High
Adult body surface area	1.8 m^2	Cumulative	Moderate
Occupational exposure	23 hr/wk	Cumulative	Moderate
Working tenure	4 yr	-	High
Time spent at home - adult	108 hr/wk	-	Moderate
Time spent away from home - adult	60 hr/wk	-	Moderate
Time spent at home - child	138 hr/wk	-	Moderate
Time spent away from home - child	30 hr/wk	-	Moderate
Residency period	8.1 yr	Cumulative	High
Time spent indoors	156 hr/wk	-	High
Time spent outdoors	12 hr/wk	-	High
Shower duration	7.6 min	-	Moderate
Soil ingestion - adult	0.1 mg/day	Cumulative	Low
Soil ingestion - child	16 mg/day	Cumulative	Moderate
Dietary intake - total	1.6 kg/day	-	Moderate
Consumption of homegrown fruits - adult	28 g/day	-	Moderate
Consumption of homegrown vegetables - adult	50 g/day	Lognormal	Moderate
Consumption of fish and shellfish - adult	14 g/day (total) 6.5 g/day (nonmarine)	Cumulative	Moderate
Consumption of beef - adult	88 g/day (total) 40 g/day (home-grown)	Normal	Moderate
Water intake - adult	1.4 L/day	Cumulative	High
Inhalation rate - adult	18 m^3/day	-	Moderate
Inhalation rate - child	12 m^3/day	-	Moderate

Source: Gephart et al., 1994

4.2.3 Chemical Manufacturers Association (Whitmyre et al., 1992a, 1992b)

Whitmyre et al. (1992a, 1992b) have summarized the results of reports published by the Chemical Manufacturers Association in which exposure assumptions used in risk assessments and the uncertainties surrounding various exposure assessment methodologies are reviewed and evaluated (CMA, 1990; CMA, 1991). Whitmyre et al., (1992a) summarize the default exposure parameters used by various federal and state agencies. Using the range of recommended values suggested by these agencies in three single-media exposure scenarios and one multimedia exposure scenario, Whitmyre et al. (1992a) found that varying only some of the parameter values could result in estimates of exposure that, in some cases, spanned as much as four-orders of magnitude. The single medium scenarios included: (1) ingestion of contaminated soil by a 2 yr-old child for one year; (2) ingestion of contaminated food (vegetables and fish) over a lifetime, and (3) dermal contact with contaminated soil over a lifetime. The multimedia scenario considered ingestion of contaminated drinking water, soil and fish. Using Monte Carlo simulations, cumulative frequency distributions of exposure were calculated for the three single-media pathways, and the results compared with estimates made using standard default value for the appropriate parameters. The results indicated that the default parameter approach resulted in exposure estimates that exceeded the 95th percentile exposure level predicted by the Monte Carlo method.

In the second report, Whitmyre et al. (1992b), present the following suggestions for reducing the uncertainties associated with estimating exposure:

- collect more extensive statistical distribution data and identify the proper distribution type for each parameter
- if defaults are used, select values that represent the central tendency (e.g., mean, 50th percentile) of the distribution
- incorporate time-activity data in the exposure/dose equations to better define appropriate exposure duration values
- use a reasonable exposure scenario, rather than a worst-case scenario
- use stochastic approaches (such as Monte Carlo simulations) and information analysis-based approaches to evaluate the variability in the exposure estimates and the likelihood or probability of each outcome.
- use bivariate analysis to identify the extent to which interdependence between different exposure parameters affects the distribution of the exposure estimates
- use less than a lifetime exposure assessment for carcinogens depending on their mode of action (initiator or promotor)
- incorporate physiological data, such as route-specific differences in absorption and pharmacokinetics, into the estimates of absorbed dose

The use of bivariate analyses is intended to reduce the variability in exposure estimates by eliminating unlikely combinations of values for parameters that are interdependent (e.g., high body weight and low ingestion rate, or low body weight and high ingestion rate).

The information analysis-based method of assessing exposure mentioned above was demonstrated by Sielken (1990). In this approach a limited number of specific alternative values are selected for each exposure parameter (i.e., distributions of discrete values). The analysis indicates the numeric value of each possible outcome and also indicates the likelihood or probability of each outcome. Therefore, the exposure outcomes can be expressed as discontinuous probability distributions made up of a specific number of discrete numeric values. The exposure estimate for a particular point in the distribution (i.e., 50th or 90th percentile) can then be used in the risk assessment.

Whitmyre et al. (1992b) note that the distribution of exposures across different pathways and routes may not necessarily reflect the distribution of risks because of the fact that absorption and target organ toxicity may be route-specific. In addition, the exposure estimates for one route may not necessarily identify the same population exposure as that for another route if different point estimates, distributions (or distribution types) are used in the calculations. Finley et al. (1994) also point out that the exposure estimates for different pathways may have different levels of uncertainty, and as a result, this may overemphasize the importance of one particular pathway where the uncertainties are highest.

4.2.4 Thompson et al. (1992)

Thompson et al. (1992) provide two hypothetical examples of using the Monte Carlo technique in health risk assessments. They estimate exposures to benzene and benzo(a)pyrene through ingestion of soil and dermal contact with soil. The distributions and point estimates used by Thompson et al. (1992) are shown in Table 4-5. The default values for exposure frequency and duration used by Thompson et al. are: 1 day/wk, 20 wk/yr, 10 yr/lifetime, for a lifetime of 70 years.

4.2.5 Paustenbach et al. (1992)

Paustenbach et al. (1992) have proposed that the most cost effective and efficient way to regulate contaminated soil is to establish "safe" concentrations for each chemical for the seven most common exposure scenarios. These exposure scenarios include (1) residential (volatiles and nonvolatiles), (2) industrial, (3) agricultural, (4) recreational, (5) groundwater, (6) wildlife and aquatic species, and (7) runoff and erosion of particulates to waterways.

Therefore, for each chemical there would be a set of seven different cleanup guidelines. Paustenbach et al. (1992) identify the most important exposure pathways for each of these scenarios and also discuss the appropriate exposure parameters to use. The default values for these parameters are shown in Table 4-6. Descriptions of the data sets available for each of the parameters and the references from which they were obtained are given by Paustenbach et al. (1992), together with explanations of how each default value was selected from the available data sets.

Table 4-5. Distribution Functions Used by Thompson et al. (1992)

Parameter	Distribution	Mean	Standard deviation[a]	Point Estimate	Location of Point Estimate	Ref.
Body weight (kg)	normal	47	8.3	47	mean	b,c
Time soil stays on skin (hr)	normal	6	1	8	95th percentile	b
Skin surface (m^2)	normal	1.4	0.17	1.4	mean	b,c
Fraction of skin exposed	lognormal	-2.15	0.5	0.2	85th percentile	b,c
Soil loading (mg/cm^2)	uniform	0.75	1.25	1	mean	b
Soil ingestion (mg/d)	lognormal	3.44	0.80	50	72nd percentile	b (d,e)

[a] For a normal, the mean and standard deviation are used to describe the distribution; for a lognormal, the mean and standard deviation of the underlying normal are used; and for a uniform distribution, the low and high values are used
[b] Calculated by Thompson et al. (1992)
[c] GCA Corporation, 1984
[d] LaGoy, 1987
[e] Thompson and Burmaster, 1991

4.2.6 McKone (1990)

An important factor in the dermal exposure/dose equation (see Section 2.5.2.12) is the chemical-specific dermal absorption factor. Experimental data on dermal absorption can be very limited and a default value of 1.0% for organic compounds has been recommended by EPA (U.S. EPA, 1992c). However, modeling studies might be able to provide a better measure of dermal absorption. McKone (1990) developed a two-layer model for estimating how chemical uptake through the stratum corneum depends on chemical properties, soil

Table 4-6. Exposure Parameters Used by Paustenbach et al. (1992)			
Parameter	**Residential**	**Occupational**	**Recreational**
Exposure duration	30 yr	25 yr	30 yr
Exposure frequency (fraction of yr)	0.962	0.343	0.011
Average lifetime	70 yr	70 yr	70 yr
Body weight	70 kg	70 kg	70 kg
Breathing rate	20 m^3/day	11 m^3/day	20 m^3/day
INGESTION OF SOIL:			
Ingestion rate	12 mg/day	10 mg/day	12 mg/day
Fraction of soil originating from source	0.47	0.5	1
DERMAL UPTAKE:			
Percent of time spent outdoors	19	100	100
Percent of time spent indoors	81	0	0
Indoor dust factor	0.75	0.75	0.75
Skin surface area	1980 cm^2	1980 cm^2	1980 cm^2
Mass of soil adhering to skin	0.5 mg/cm^2	0.5 mg/cm^2	0.5 mg/cm^2
Fraction of soil originating from source	0.5	1	0.5
INHALATION OF PARTICULATES:			
Concentration of total suspended particulates	0.25 mg/m^3	0.1 mg/m^3	0.1 mg/m^3
Fraction of soil originating from source	0.47	0.5	1
Fraction deposited to lung tissue	0.25	0.25	0.25
Percent of time spent outdoors	19	100	100
Percent of time spent indoors	81	0	0
Indoor dust factor	0.75	0.75	0.75

Source: Paustenbach et al., 1992

properties, and exposure conditions. McKone found that the fraction of soil-bound chemical that passes through the stratum corneum is dependent on the skin-soil layer thickness, the dimensionless Henry's Law constant, K_h, and the octanol-water partition coefficient, K_{ow}, of the soil-bound chemical. McKone used a number of default values in his model for soil and skin characteristics including those listed in Table 4-7. The model predicted that uptake

Table 4-7. Exposure Parameters Used by McKone (1990)		
Parameter	**Values**	**Units**
Exposure duration	12	hr
Skin area exposed	0.5	m^2
Octanol-water partition coefficient	10^3 - 10^7	-
Air-water partition coefficient	10^{-6}, 10^{-3}, 1	-
Diffusion coefficient in air	5×10^{-6}	m^2/sec
Diffusion coefficient in water	5×10^{-10}	m^2/sec
Soil bulk density	1600	kg/m^2
Soil porosity	0.5	-
Soil water content	0.3	-
Soil air content	0.2	-
Soil total density	1900	kg/m^2
Soil organic carbon fraction	0.02	-
Stratum corneum thickness	1.5×10^{-5}	m
Fat content of stratum corneum	0.1	kg/kg
Water content of stratum corneum	0.5	m^3/m^3
Density of skin	1000	kg/m^3
Air side boundary layer	0.005	m
Mass of soil deposited on skin	5×10^{-4} to 0.5	kg
Mass of soil per unit area on skin	0.1 to 100	mg/cm^2

Source: McKone, 1990

would be strongly dependent on the amount of soil on the skin surface. For amounts less than 1 mg/cm^2, the model predicted rather large uptake rates, in some cases approaching unity. Although McKone states that the model should not be applied to all human exposure models until the variation of uptake fraction with soil loading can be evaluated, he suggests that the following generalizations might be valid: for compounds with a K_{ow} of 10^6 or below and a K_h below 0.001, "it is not unreasonable to assume 100% uptake in 12 hr. Second, for compounds with a K_h of 0.01 and above, the uptake fraction is unlikely to ever exceed 40% in 12 hr, and will be well below this when K_{ow} is above 10. Third, for compounds with a K_h of 0.1 and above, we can expect no more than 3% uptake in 12 hr".

Burmaster and Maxwell (1991) evaluated the time-loading dependence in the McKone model for monocyclic and polycyclic aromatic hydrocarbons and found that as K_{ow} increases and as K_h decreases, the transfer kinetics slow and the soil layer contributing molecules to dermal uptake becomes thinner.

4.2.7 Maxim (1989)

Maxim (1989) points out that one factor often overlooked in exposure assessments is pollutant depletion/removal over time through processes such as volatilization, erosion, dispersion/dilution, photolysis, chemical reaction, and biodegradation. Assuming first-order kinetics, Maxim calculated the average contaminant concentration, as a fraction of the initial concentration, for different rates of degradation (half-lives) and for different time periods (1-40 yr). An abbreviated version of the resulting calculation is shown in Table 4-8. Since exposure and dose is proportional to level of contamination, the relative decrease in the concentration provides an indication of the decrease in risk. For example, for a contaminant with a half-life of 10 years, the average contaminant concentration beginning after a 5-yr occupancy lag would be about 17% of the initial value.

Table 4-8. Average Contaminant Concentration as Fraction of Initial Value[a,b]

Occupancy Lag (years)	Half-life of Pollutant (years)				
	1	2	5	10	20
	First-order Kinetic Constant (1/yr)				
	0.693	0.347	0.139	0.00693	0.00347
1	1.05E-02	2.96E-02	9.10E-02	1.93E-01	3.67E-01
3	2.69E-03	1.52E-02	7.10E-02	1.73E-01	3.50E-01
5	6.94E-04	7.85E-03	5.55E-02	1.55E-01	3.34E-01
7	1.79E-04	4.05E-03	4.34E-02	1.39E-01	3.19E-01
10	2.35E-05	1.50E-03	3.01E-02	1.18E-01	2.98E--01
16	4.08E-07	2.09E-04	1.45E-02	8.61E-02	2.60E-01
20	2.75E-08	5.64E-05	9.01E-03	6.99E-02	2.38E-01

[a] Adapted from Maxim, 1989.
[b] First-order removal with exposure endpoint of 70 years.

Another suggestion by Maxim (1989) is that the exposure assessment incorporate consistency and plausibility audits into modeling the exposure scenario to avoid developing unrealistic situations (such as where the exposure/dose for the entire expected exposure exceeds the amount actually present). Maxim also supports the use of sensitivity analyses as an integral part of exposure assessments.

4.2.8 Driver et al. (1989)

Driver et al. (1989) reported that adherence of soil to skin was significantly correlated with soil particle size, and suggested that site-specific soil adherence values, based on weighted average values according to the particle size distribution at the specific site, be used in dermal exposure scenarios. They also suggest that site-specific soil organic content be used to evaluate dermal absorption.

4.2.9 McKone (1988a, 1988b; McKone and Daniels, 1991; Daniels and McKone, 1991)

McKone (1988, see also McKone and Daniels, 1991) developed a set of multimedia, multipathway models for estimating potential human exposure to environmental contaminants. The models link contaminant concentrations in air, water, and soil to human exposure through inhalation, ingestion, and dermal contact with the use of pathway exposure factors (PEFs). The PEFs are algebraic expressions that link exposure parameters with models of environmental partitioning and translate contaminant concentrations into lifetime chronic daily intakes (CDI) normalized to body weight (i.e., mg/kg-day). The PEFs, therefore, are a measure of exposure (per unit body weight) estimated from the partitioning of the contaminant between successive environmental media. Total exposure through multiple media and multiple pathways can be obtained by summing the exposures for all relevant pairs of media (i.e., for ingestion of meat, the pathways that would be considered would be soil to pasture, pasture to cattle, and cattle to humans). Nine exposure pathways (inhalation, dermal contact, and ingestion of water, fruits and vegetables, grains, meat, fish, and soil) and five environmental media (volatiles in air, particles in air, soil, drinking water, and surface water) are evaluated and 23 specific PEFs are calculated by McKone and Daniels (1991). The contaminant concentrations in the media were estimated using a multimedia transport and transformation model, such as GEOTOX (McKone and Layton, 1986; McKone et al., 1987).

The general exposure/dose equation used by McKone is as follows:

$$CDI = F_{ki} \times C_k \qquad (4\text{-}4)$$

where:

CDI = Chronic daily intake (mg/kg-day)
F_{ki} = Pathway exposure factor (PEF) for medium k and pathway i (L/kg-day)
C_k = Concentration in the environmental medium k (mg/L, for drinking water)

The F_{ki} (PEF) is derived as follows:

$$F_{ki} = \left[\frac{CR_i}{BW}\right] x \frac{C_i}{C_k} x \frac{EF \times ED}{AT} \quad \text{(4-5)}$$

where

CR_i = Contact (intake) rate for pathway i (units specific for each pathway)
BW = Body weight (kg)
C_i = Contaminant concentration in exposure pathway i (units specific for each pathway)
C_k = Contaminant concentration in environmental medium k (units specific for each pathway)
EF = Exposure frequency (days/yr)
ED = Exposure duration (yr)
AT = Averaging time (days)

Assuming that environmental concentrations are constant over time, the lifetime-equivalent CDI is derived by averaging exposures over all age groups in a population. The PEFs derived by McKone are dependent on the contact or intake rates used in the calculations. These values are shown in Table 4-9. The values for body weight, surface area, breathing rates, and fluid intake were obtained from data published by the International Commission on Radiological Protection (ICRP, 1975). Food consumption rates were obtained using the results of the statistical analysis conducted by Yang and Nelson (1986) on survey information compiled by the U.S. Department of Agriculture (USDA, 1983).

4.2.10 Hoffman et al. (1993)

Hoffman et al. (1993) recommend that the initial phase of a risk assessment for a Superfund site be a screening analysis involving two separate sets of calculations designed to provide upper and lower bound estimates of health risks. The upper bound estimate of risk is derived using maximum contaminant concentrations and conservative default values for the exposure parameters. The lower bound estimate of risk is derived using the mean or geometric mean contaminant concentration and nonconservative parameter values which, in some cases, are a factor of 10 less than the reasonable maximum estimates. The resulting exposure (dose) estimates are then divided by the RfD for noncarcinogens, or multiplied by the slope factor (carcinogens) to develop a screening index for the contaminant.

Table 4-9. Exposure Parameters Used By McKone and Daniels (1991)[a]			
Parameter	Child[b]	Adult[c]	Combined[d]
Body weight (kg)	27	66	58
Surface area (m^2)	0.95	1.7	1.5
Active breathing rate (m^3/kg-hr)	0.030	0.018	0.021
Resting breathing rate (m^3/kg-hr)	0.011	0.0060	0.0070
Fluid intake (L/kg-day)	0.05	0.03	0.034
Fruit and vegetable intake (kg/kg-day)	0.008	0.0045	0.0053
Grain intake (kg/kg-day)	0.0074	0.0030	0.0040
Milk intake (kg/kg-day)	0.014	0.0033	0.0056
Meat intake (kg/kg-day)	0.0044	0.0028	0.0031
Fish intake (kg/kg-day)	0.00037	0.00030	0.00030
Soil ingestion (mg/kg-day)	4.3	0.71	-

Source: McKone and Daniels, 1991
[a] Arithmetic mean values (estimated standard deviations are also available in the report)
[b] Ages 0-15
[c] Ages 16-70
[d] Derived as the sum of the child's value multiplied by 15/70 and the adults value multiplied by 55/70

4.2.11 American Industrial Health Council (AIHC, 1994)

The American Industrial Health Council (AIHC, 1994) has summarized and evaluated the current scientific documentation and statistical data for various exposure parameters used in risk assessments. These include: (1) contact rates with environmental media (e.g., ingestion rates of water, food and soil); (2) exposure frequency and duration (e.g., length of time at one residence, time spent outdoors, time spent at home); (3) body weight; (4) averaging time; and (5) chemical specific factors (e.g., transport through the lungs, skin and gut). For each parameter a rationale for use of a specific point value is provided which is considered to be the best single value to characterize typical populations. When available, information is also included by the range of values in existence for the exposure factors as well as statistical treatment of the parameter distribution data. The reference values derived by AIHC are shown in Table 4-10 (Note: these values appear to be the same as those listed by Gephart et al., 1994).

Table 4-10. Reference Values Recommended by AIHC (1994)			
Parameter	Point value	Recommended Distribution	Data quality
Adult body weight	72 kg	Cumulative	High
Child body weight	13 kg	Cumulative	High
Adult male body surface area	1.93 m^2	Cumulative	Moderate
Adult female body surface area	1.69 m^2	Cumulative	Moderate
Time at work	23 hr/wk	Cumulative	Moderate
Working tenure	4 yr	-	High
Time spent at home - adult	108 hr/wk	-	Moderate
Time spent away from home - adult	60 hr/wk	-	Moderate
Time spent at home - child	138 hr/wk	-	Moderate
Time spent away from home - child	30 hr/wk	-	Moderate
Residency period	8.1 yr	Cumulative	High
Time spent indoors	156 hr/wk	-	High
Time spent outdoors	12 hr/wk	-	High
Shower duration	7.6 min	Cumulative	High
Soil ingestion - adult	0.1-10 mg/day[a]	Cumulative	Low
Soil ingestion - child	16 mg/day	Cumulative	Moderate
Dietary intake - total	1.6 kg/day	-	Moderate
Intake of homegrown fruits - adult	28 g/day	-	Moderate
Intake of homegrown vegetables - adult	50 g/day	Lognormal	Moderate
Intake fish/shellfish, total - adult	14 g/day	Cumulative	Moderate
Intake of nonmarine fish/shellfish - adult	6.5 g/day	Cumulative	Moderate
Intake of beef, total - adult	88 g/day	Normal	Moderate
Intake of beef, homegrown - adult	40 g/day	Normal	Moderate
Water intake - adult	1.4 L/day	Cumulative	High
Inhalation rate - adult	18 m^3/day	-	Moderate
Inhalation rate - child	12 m^3/day	-	Moderate

Source: AIHC, 1994

[a] A range of values is recommended over a point value because of the shortage of reliable data.

5. KEY ISSUES

This section summarizes key issues that have been identified in this report. Two issues stand out from the rest; (1) a possible change in the way that contaminant concentrations used in exposure assessments are derived (see Section 5.3), and (2) the use of distributions of parameter values (instead of default values) and stochastic methods such as Monte Carlo simulations to estimate population exposure (see Section 5.5). Both represent major changes in the way exposure assessments are conducted. The first of these, which has been recommended by EPA's Science Advisory Board (SAB), would take into account the spatial distribution of contaminant concentrations in soil. The second, which has received endorsement by both SAB and the National Research Council (NRC), would generate probability distributions for population exposure. Exposure values could then be identified for specific percentiles of the population to define acceptable levels of protection for the population that is or may be exposed.

Although this report has focused on federal exposure assessment guidelines, brief summarys have also been given of some, but not all state guidelines. Because individual state regulations may, under some circumstances, supersede CERLA requirements for establishing cleanup standards, a more detailed evaluation and comparison of state guidelines for risk assessments may be warranted.

5.1 SOURCE ASSESSMENT

The first steps in the exposure assessment process involve the characterization of the site and the identification of the contaminants. General information on the physical characteristics of the site and on the physical/chemical properties of the contaminants can be useful in streamlining the source assessment and identifying those media or exposure pathways that do or do not require evaluation. For example, in proposing generic soil cleanup guidelines for 2,3,7,8-tetrachlorodibenzo-*p*-dioxin (TCDD), Paustenbach et al. (1992) note that the chemical and physical properties of TCDD "show that it is essentially immobile in soil and virtually not water-soluble, so groundwater is unlikely to be threatened by TCDD. The only exception might be in those situations where the TCDD is present (in solution) in solvents in soil". Similarly, information on the volatility of contaminants might be used to screen inhalation pathways, and information on octanol-water partitioning coefficients might be used to screen dermal contact pathways. A standardized evaluation and interpretation of key physical/chemical and environmental fate parameters (i.e., water solubility, lipid solubility, soil adsorption constant, vapor pressure, bioconcentration factor, and soil half-life), as well as key site parameters (i.e., type of soil, depth of groundwater, amount of rainfall, etc.), could be very useful in setting priorities in a risk assessment.

5.2 ENVIRONMENTAL PATHWAYS AND FATE

The movement of contaminants between environmental media can be evaluated using environmental fate and process models. Such models can also be used to derive partitioning coefficients. Partitioning coefficients, in turn, can be used to estimate exposures through multiple environmental media. In addition, the output of such models can be used as a screening device in risk assessments. EPA has proposed soil-screening levels derived from environmental fate models using "conservative" default exposure parameters and specific exposure scenarios (U.S. EPA, 1994d). Contaminant concentrations at a particular site that do not exceed the screening levels would be considered safe for most uses. The NRC supports a tiered approach to risk assessments in which the first step would involve screening procedures. Further development and standardization of such a methodology has the potential for simplifying and accelerating the risk assessment process.

5.3 ESTIMATING CONTAMINANT CONCENTRATIONS

For Superfund risk assessments, EPA recommends using the 95% upper confidence limit (UCL) on the arithmetic mean contaminant concentration. EPA's Science Advisory Board (SAB) has given several reasons why the use of the UCL and/or the arithmetic mean would be inadequate for calculating the reasonable maximum exposure (RME), particularly for exposures to contaminated soil (see section 3.1). SAB states that the use of the mean concentration is appropriate primarily for estimating population risks and only if spatial and temporal variability in the concentrations is small and potential exposures are equally likely across all parts of the site. SAB recommends instead the use of either (1) an average contaminant concentration weighted by the likelihood of the sampling site being visited or (2) a specific percentile of a distribution of average concentrations to which visitors to the site might be exposed. Furthermore, SAB recommends the use of statistical approaches, such as kriging and triangulation, which would take into account the spatial distribution of soil contaminants across the site.

Potential exposure concentrations may change over time due to changing environmental conditions, flux of contaminants between environmental media, and natural rates of physical, chemical and biological degradation. Some investigators have suggested that a key step in the exposure assessment process be an evaluation of these possible changes. Models which estimate rates of biological and physio-chemical degradation of contaminants in soil can be used to predict future changes in contaminant concentrations. The calculation of time-weighted average exposure concentrations resulting from natural degradation processes would provide an estimate of exposure that would be less conservative than estimates based on current fixed concentrations in bulk media. For exposure

assessments at a particular site, possible changes in contaminant concentrations with time might also be evaluated with consideration given to the length of time that the site might remain unoccupied and/or unused.

Exposures derived from models may predict concentrations below current detection limits, but above reference toxicity values. The applicability of such estimates therefore depends largely on the accuracy and reliability of the models used. To reduce the uncertainty surrounding the use of models, EPA suggests that different modeling formulations be used and the range of outputs be considered to be representative of the range in uncertainty.

5.4 POPULATION ANALYSIS

5.4.1 Exposure Scenarios

Although the Superfund Guidelines (U.S. EPA, 1989a) allow for the evaluation of a number of exposure scenarios, some investigators have suggested that for the sake of consistency between state and federal agencies, several exposure scenarios should be selected and standardized (i.e., using an agreed upon set of assumptions or distributions for exposure parameters) so that "safe" environmental concentrations can be identified for these specific scenarios. Paustenbach et al. (1992) have proposed seven scenarios including (1) residential (volatiles and nonvolatiles), (2) industrial, (3) agricultural, (4) recreational, (5) groundwater, (6) wildlife and aquatic species, and (7) runoff and erosion of particulates to waterways. Standardization of exposure scenarios, even if adopted for screening purposes alone, may provide benefits in terms of time and cost of doing preliminary risk assessments for specific sites.

Not all possible exposure scenarios are addressed by the Superfund guidelines; consequently, default values for exposure parameters used in other scenarios may not be available. For example, there are no guidelines for evaluating exposure to the inadvertent or intentional trespasser to a site. Exposure scenarios which may be relevant for ERAP sites to be identified (i.e., military or limited scientific use), and the appropriate exposure pathways and parameters values for those scenarios need to be defined.

5.4.2 Exposure Parameters and Pathways

The exposure/dose equations shown in Section 2.5.1, contain exposure parameters that are spatially or temporally variable. For example, contaminant concentrations in environmental media may change due to abiotic (weather) or biotic factors (biodegradation).

Population-specific factors such as intake rates may vary among individuals within the population and may also change over time (i.e., age-specific changes). Much work has been done in characterizing the range in variability of population-specific exposure parameters. Standardization and continuous updating of the data sets, based on the most recent studies, would be desirable to provide more reliable estimates of exposure.

For those parameters that vary with age, time-weighted average values have been used in some types of assessments, such as in developing soil screening levels (U.S. EPA, 1994d). Further consideration might be given as to whether this approach should be applied in baseline risk assessments.

Several researchers have undertaken sensitivity analyses to determine how current limitations on information relating to distributions of exposure parameters affect estimates of exposure. The suggestion has been made that this be done on a systematic basis to identify those parameters that have the greatest impact on uncertainty surrounding the exposure estimates, and that additional research be conducted to better define the distributions of those parameters.

Soil Ingestion - The NRC guidelines recommend that EPA conduct pediatric risk assessments whenever children may be at greater risk than adults. The EPA methodology does identify children 1-6 years old as a high risk group associated with ingestion and dermal contact with contaminated soil. In Superfund risk assessments, exposures through soil ingestion for this age group are compared to an RfD calculated for a chronic (70-yr) exposure. However, in the methodology for developing Preliminary Remediation Goals, EPA uses a time-weighted average soil ingestion rate for a 30-year exposure (6 years for children and 24 years for adults). EPA's Science Advisory Board has stated that a time-weighted exposure that is compared to a chronic RfD appears to be the preferred option; however, it recommends that both options (as well as one in which children's 6-year exposure is compared with a subchronic RfD) should be further evaluated. Specific EPA guidance on the assessment of risks to children with pica is not available.

Information on soil ingestion rates for adults is very limited, and currently used defaults may overestimate this value for residential scenarios. New data or a more detailed evaluation of the currently available data might indicate that this pathway accounts for such a small fraction of the overall exposure at most sites that it could be excluded from most risk assessments.

Adjustments for Absorption Efficiency - Within a single exposure pathway, adjustments to the dose estimation may be needed if different absorption efficiencies occur within different media. For example, contaminants ingested with food or soil may be less completely

absorbed than those ingested with water. In the absence of experimentally derived data, the Superfund Guidelines indicate that the assumption should be made that relative absorption efficiencies between food, water and soil equal 1.0.

In evaluating exposures resulting from ingestion of contaminated soil, EPA recognizes that the bioavailability of the contaminant may be affected by the adsorption of the contaminant to the soil matrix. Some methodologies allow for this possibility into their exposure/dose equations. In the absence of experimental data, the default assumption is that the contaminant is 100% bioavailable for absorption through the gastrointestinal tract. Modeling efforts based on the physical/chemical properties of the contaminant and the soil, might be developed to better define the effective dose.

Dermal Exposures - The dermal exposure/dose equation presented in the Superfund Guidelines incudes a chemical-specific dermal absorption factor. EPA does not have a standardized method for determining rates of dermal absorption, although default values have been recommended by Regional Offices (e.g., Region IV has recommended bioavailability factors of 1.0% for organic compounds and 0.1% for inorganic compounds in soils, these values take into account binding of the substance to the soil matrix). The state of Massachusetts recommends the following default values for dermal absorption: 10-25% for VOCs; 1-10% for SVOCs; 1-10% for pesticides, and 0.1-1% for inorganics (MDEQE, 1989). Recent studies suggest that under certain conditions dermal absorption may approach 100% for some organic compounds (McKone, 1990). Dermal uptake models have been included in dermal exposure equations used by some researchers. Standardization of modeling techniques for dermal absorption could provide better estimates of dermal exposures.

For evaluating dermal uptake of airborne or aqueous chemicals the permeability coefficient of the contaminant is used; however, for dermal uptake of contaminants from soil a dermal absorption factor approach is used. Permeability coefficients are independent of the concentration of the chemical in the medium, whereas the absorption fraction varies with the concentration. EPA's Dermal Exposure Assessment guidelines provide an number of alternatives for estimating dermal permeability coefficients and dermal absorption factors. Development of standardized permeability coefficients for soil contact scenarios might reduce the uncertainties surrounding this parameter.

In the absence of experimental data on dermal absorption rates, consideration might be given to using indirect methods for estimating absorption. The evaluation of the same endpoint for intravenous and dermal routes of exposure in an animal species (i.e., LD_{50} or ED_{50} values) could provide a very approximate estimate of dermal absorption. For example, if the dermal LD_{50} is 5 times greater than the intravenous LD_{50}, then the assumption might be made

that only 20% of the dermal dose is absorbed. An evaluation would have to be made as to whether such estimates would also apply to long-term exposures to low doses, and also whether the same relationship seen in an animal study would be true for humans.

Multipathway Exposures - In determining total multipathway exposure to a specific contaminant, the single pathway exposures must be evaluated using the same dose derivation, i.e. either administered or absorbed dose. Dermal exposures are calculated as absorbed doses and ingestion doses are calculated as administered doses. The latter are converted to absorbed doses using chemical- and pathway-specific absorption factors (i.e., percent of chemical absorbed). In the absence of experimentally derived absorption factors, default values must be used. The Superfund Guidelines do not provide gastrointestinal default values for organic compounds, but they do indicate that a default value of 5% would be a relatively conservative assumption for metals. EPA Region IV has adopted as interim default values for gastrointestinal absorption a value of 80% for volatile organics, 50% for semivolatile organics, and 20% for inorganics. Development of more refined estimates of absorption factors, such as for specific chemical classes, would reduce the uncertainties surrounding this parameter and provide better estimates of total multipathway exposure.

Combining an exposure from an inhalation pathway with doses derived from other pathways requires the conversion of an atmospheric concentration to an administered or absorbed dose. In Superfund risk assessments, this is accomplished using a set of default assumptions for an adult male (i.e, inhalation rate of 20 m^3/day and a standard body weight of 70 kg), and in the absence of any specific data, a pulmonary absorption factor of 100% is assumed. However, if the exposure assessment focuses on multipathway exposure to a population that is different in inhalation rate or body weight, such as women or children, then these default values would not be applicable. There is no EPA guidance as to whether in such cases the inhalation exposure calculation should be adjusted using different default parameters.

5.5 INTEGRATED EXPOSURE/DOSE ANALYSIS

If exposure/dose can be calculated for each individual in an exposed population, based on that individual's unique exposure history (i.e., intake and contact rate, and contaminant concentration), the result would be a range of exposures which could then be used to describe the entire exposed population in terms of the average, minimum, and maximum exposure. In addition, cumulative frequency distributions could be used to identify exposures corresponding to the median, 90th, 95th or some other percentile level. However, in the absence of information on individual exposures, and in the estimation of potential exposures for a future-use scenario, parameter values are needed that are representative of either specific segments

of the population or of the population as a whole. Therefore, a key issue in exposure assessments is in selection of values for the parameters used in the exposure/dose equations.

Selection of exposure/dose values - EPA currently recommends using three sets of exposure/dose values in risk assessments; bounding estimates, central tendency (CT) and high-end exposure estimates (HEEE, equivalent to reasonable maximum exposure, RME). Bounding estimates are used only as screening levels. The CT integrates exposures across all individuals, and the HEEE (RME) focuses on the upper percentiles of the exposed group (U.S. EPA, 1992a). This procedure is in keeping with the recommendations of the NRC (1994) that a tiered approach be used in assessing exposure and risk. EPA's Science Advisory Board has also recommended that a "most reasonable exposure" be calculated together with an RME (U.S. EPA, 1993j). EPA states that in developing a HEEE, the exposure parameters should not be selected so that all are at the high end of their range (U.S. EPA, 1992a). This is to avoid deriving an exposure estimate that is outside the normal range for the population. NRC recommends that methodology be adopted so that HEEE is consistently representative of a specific subgroup, based on the size of the exposed population.

It has been suggested by various groups, including EPA's Science Advisory Board, that exposure estimates be based not on point values for the exposure parameters, but instead on probability density functions (PDF) (or equivalently cumulative frequency distributions). PDFs could then be used in stochastic analyses such as Monte Carlo simulations to estimate population exposures, and specific points on the exposure distribution curve could then be used as the CT and HEEE. Standardization of the data sets and types of distribution that should be used for each parameter (i.e., normal, lognormal, etc.) would contribute substantially in improving the overall exposure assessment process.

As noted by EPA and other researchers, stochastic simulations such as the Monte Carlo technique may produce implausible exposure distributions if open-ended parameter distributions (such as lognormal distributions) are used in the calculation. When insufficient data are available, it has been recommended that the parameter distributions be truncated at the 1st and 99th percentile values or at the 5th and 95th percentile values (depending on the statistical data available for the given parameter). Uncertainties will also be incorporated in stochastic simulations if interdependent factors, such as age, body weight and food intake, are not taken into account.

Tiered Approach - NRC (1994) has recommended that EPA use an tiered approach in conducting risk assessments such that the first stage would involve the use of screening procedures which would eliminate low priority chemicals from further consideration and thereby allow the risk assessment to focus on high priority chemicals. EPA's development of soil screening values (based on the methodology for deriving Preliminary Remediation Goal)

is a step in this direction (U.S. EPA, 1994d, g,h). Once developed, screening values can be used across sites, thereby making the risk assessment process more efficient and less costly.

Uncertainty Analysis - NRC (1994) has recommended that EPA incorporate uncertainty analysis into its exposure and risk assessments. EPA (1992a) has recognized this need and supports the use of probabilistic techniques such as the Monte Carlo simulations, so long as the limitations of the methods are recognized and considered in the overall risk assessment process. Because there are a number of simulation programs available (i.e., Monte Carlo, Latin Hypercube analysis), criteria may need to be standardized for choosing one method over another.

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APPENDIX A

SOIL EQUILIBRIUM PARTITIONING MODEL

A simple model to predict partitioning of a chemical between the soil, soil-water and soil-air in a soil/ground-water system is shown below (adapted from USAF, 1989). Many other models are available and this example is only for illustrative purposes.

The general approach used in this model is that of a Level I model of Mackay (1979) and MacKay and Patterson (1981, 1982). The key assumptions are:

- equilibrium partitioning
- no degradation of the chemical takes place
- all portions of the compartments (soil/water/air) are equally accessible
- the ground water is relatively clean, fresh water
- linear adsorption isotherm for soil sorption
- for topsoils, sorption is due to physical interaction with the organic fraction of soil
- the chemical is present as a trace contaminant (below the solubility limit in water)

A model topsoil (unsaturated) was defined having the following characteristics:

- Soil volume = 60%
- Air volume = 30%
- Total porosity = 40%
- Water volume = 10%
- Organic carbon content of soil = 2% (by weight)
- Bulk density of soil = 1.6 g/cc.
- Temperature = 20°C or 25°C

There are only two chemical-specific inputs required by the model:

(1) H: Henry's law constant (atm • m^3/mol)
(2) K_{oc}: Soil sorption constant per unit weight organic carbon in soil (no units)

At 20°C, the three model-derived equations for the fraction, F, in each soil compartment are:

$$F\ (air) = 12.5/[12.5 + (0.1 + 0.0192\ K_{oc})/H]$$
$$F\ (water) = 0.1/H[12.5 + (0.1 + 0.0192\ K_{oc})/H]$$
$$F\ (soil) = 0.0192K/H\ [12.5 + (0.1 + 0.0192\ K_{oc})/H]$$

Note that F(air) + F(water) + F(soil) = 1.
For a calculation at 10°C, change all "12.5" values to "12.9".

A model deep-soil (saturated) was defined having 30% (by volume) water and 70% soil with 0.1% by weight organic carbon. This small amount of organic carbon was assumed (unless otherwise noted) to be responsible for the extent of the chemical's sorption. The bulk density of the soil was taken as 1.8 g/cc. The equations for the fractions associated with the water and soil phases are then calculated from:

$$F(water) = 0.3/[0.3 + 1.26E\text{-}03\ K_{oc}]$$
$$F(soil) = 1.26E\text{-}03\ K_{oc}/[0.3 + 1.26E\text{-}03\ K_{oc}]$$

EXAMPLE:

For di(2-ethylhexyl)phthalate, the model predicts a low potential for groundwater contamination (see table below); however, it does not take into account the possibility of the formation of water-soluble complexes with organics such as fulvic acid. The latter may enhance transport of the chemical through the soil to groundwater.

EQUILIBRIUM PARTITIONING OF DI(2-ETHYLHEXYL)PHTHALATE IN SOIL[a]

K_{oc}[b]	Depth	Estimated Percent in Each Compartment[c,d]		
		Soil	Soil water	Soil-air
62,100	Unsaturated topsoil (20°C)	99.99	0.008	<0.0001
62,100	Saturated deep soil	99.6	0.4	-

[a] Calculations based on Mckay's model (Mackay, 1979; Mackay and Patterson, 1981,1982) using assumptions and limitations as discussed in USAF (1989).
[b] K_{oc} value derived from log K_{oc} = log K_{ow} - 0.317 (Means et al., 1982).
[c] Henry's Law Constant of 2.5 x 10^{-7} atm-m^3/mol was used in the calculation.
[d] Calculated percentages should be considered only as a rough estimates.

APPENDIX B

ENVIRONMENTAL PARAMETERS WHICH MAY BE USEFUL FOR SCREENING PURPOSES

Atmospheric Half-life ($t_{½}$): The following reactivity scale has been suggested for organic compounds based on the hydroxyl radical rate constants and half-lives (Darnall et al., 1976):

Class	Half-life	Reactivity
I	>9.9 days	<10
II	24 hr to 9.9 days	10-100
III	2.4-24 hr	100-1000
IV	0.24-2.4 hr	1000-10,000
V	<0.24 hr>	10,000

where: half-life = $0.693/k_{OH}[OH]$ and reactivity relative to methane = 1

Bioconcentration Factor (BCF): the BCF is the equilibrium ratio of the concentration of a chemical in an exposed organism to the concentration of the chemical in the surrounding water. For organic compounds, BCFs range from about 1 to 1,000,000 (see Lyman et al., 1982). An approximate indication of relative bioconcentration potential of a chemical substance can be made by dividing this range into three categories:

low potential	<10
moderate potential	10-1,000
high potential	>1,000

Biodegradation: the transformation of chemical compounds primarily by microorganisms (e.g., bacteria, fungi) in soil and water. Biodegradation can be expressed in terms of a rate constant (k_{bio}) and half-life ($t_{½}$) for those processes following first order kinetics. Based on half-life, the relative biodegradability of a compound can be indicated as follows:

negligible	half-life, >months
slow	half-life, months
moderate	half-life, weeks
rapid	half-life, ≤ days

BOD/COD Ratio: The ratio of the biological oxygen demand (BOD) to the chemical oxygen demand (COD) which can provide a measure of the degradability of a compound (Lyman et al., 1982):

relatively undegradable	BOD/COD < 0.01
moderately degradable	BOD/COD 0.01-0.1
relatively degradable	BOD/COD >0.1

Direct aqueous photolysis rate constant (k_d): the rate constant (in $mol \cdot L^{-1} \cdot yr^{-1}$) for the direct photolytic transformation of an organic compound in water.

Henry's Law Constant (H_c): the air/water partition coefficient, usually estimated by dividing the vapor pressure of a sparingly water soluble chemical substance by its water solubility. H_c provides a measure of the volatility of the chemical from water or soil. For organic compounds, H_c (in $atm\text{-}m^3/mole$) values range from $< 10^{-7}$ to about 10^{-2} $atm\text{-}m^3/mole$ (see Lyman et al., 1982). The relative volatility of a compound from water varies in the following way:

essentially nonvolatile	$H \leq 3 \times 10^{-7}$ $atm\text{-}m^3/mole$
slow volatilization	$H \sim 10^{-7}$ to 10^{-5} $atm\text{-}m^3/mole$
significant volatilization	$H > 10^{-5}$ to 10^{-3} $atm\text{-}m^3/mole$
rapid volatilization	$H > 10^{-3}$ $atm\text{-}m^3/mole$

Hydrolysis: the transformation process in which a molecule, abbreviated RX, reacts with water, forming a new chemical bond between R and oxygen derived from water, and cleaving the bond between R and X. Relative rates can be expressed in terms of half-life.

negligible	half-life, >years
slow	half-life, months
moderate	half-life, days
rapid	half-life, ≤ minutes

Hydroxyl radical rate constant ($k_{OH\cdot}$): the rate constant (in $cm^3 \cdot mol^{-1} \cdot sec^{-1}$) for the reaction of photochemically produced hydroxyl radicals with organic compounds in the atmosphere.

Ionization or Acid Dissociation Constant (K_a): an equilibrium ratio of the dissociation products and the parent compound in aqueous solutions. The degree of dissociation can alter the solubility and adsorption characteristics of the compound. The pK_a is the negative log of K_a. If the pK_a of an organic acid, is equal to the pH of an aquatic system, 50% of the compound will exist in its anionic form (A^-); if pK_a-pH = 1, only 9.1% of the compound will be dissociated, and if pK_a-pH = -1, then 90.0% of the compound will be dissociated (Bacci, 1993). Therefore, organic acids with pK_a values equal to or less than the pH of natural aquatic systems will exhibit significant dissociation ($\geq$50%). In contrast, for basic compounds, dissociation increases with increasing pK_a.

Octanol-Water Partition Coefficient (K_{ow}): the equilibrium ratio of a chemical's concentration in the octanol phase to its concentration in the aqueous phase of a two-phase octanol-water system, typically expressed in log units (log K_{ow}). K_{ow} can be used to estimate the propensity of a chemical to be bioconcentrated in aquatic organisms, and also sorbed to the organic components of soils and sediments. Log K_{ow} values for a variety of organic compounds range from about -2 to >7 (see Verschueren, 1983). An approximate indication of relative potential for a compound to partition to lipids can be made by dividing this range into three categories:

low potential	log K_{ow} = <0	K_{ow} = <1
moderate potential	log K_{ow} = 0-3	K_{ow} = 1-1000
high potential	log K_{ow} = >3	K_{ow} = >1,000

Photolysis: the transformation of a chemical by light energy. The potential for direct photochemical transformation exists if absorption occurs in the UV spectrum > 290 nm (Radding et al., 1977).

Plant Uptake: the uptake of a chemical into plants is expressed in terms of a bioconcentration factor (BCF) which is the ratio of the concentration in the plant tissue to the concentration in soil. Generally, chemicals with K_{ow} values less than about 2 will have BCF values 1-2; those with K_{ow} values 2-3 will have BCF values around 1; and those with K_{ow} about 3-5 will have BCFs of 1-2.5 (Trapp, 1993). Chemicals with very high K_{ow} (>5) tend to be very strongly bound to soil and are only very slowly taken up by plants.

Refractory Index (RI): the ratio of the ultimate biochemical oxygen demand (BOD_u) to the ultimate oxygen demand (UOD), indicating the proportion of the theoretical total oxidation of an organic compound that is attributed to microbial action. An RI approaching 1.0 indicates that a substance is readily degraded to the point of mineralization. An error factor of about 13% is associated with estimates of RI (Lyman et al., 1982).

Soil Half-life ($t_{½}$): Relative persistence of chemicals in soil can be indicated by half-life. Mauskopf and Curtis-Powell (1985) suggest the following scale:

low persistence	$t_{½}$ = <60 days
moderate persistence	$t_{½}$ = 60-240 days
high persistence	$t_{½}$ = >240 days

Soil-Organic Carbon Sorption Coefficient (K_{oc}): a measure of the extent to which a chemical partitions between the solid and solution phases of a two-phase system (soil, sediment or activated sludge) expressed on an organic carbon basis. The equilibrium ratio of the amount of chemical sorbed per unit weight of organic carbon (oc) to the concentration of the chemical in solution. For organic compounds, soil adsorption coefficients (K_{oc}) range from 1 to 10,000,000 (see Lyman et al., 1982). An approximate indication of relative soil adsorption potential of organic compounds can be made by dividing this range into three categories:

low potential	K_{oc} = 1 to 100	log K_{oc} = 0-2
moderate potential	K_{oc} = 100 to 10,000	log K_{oc} = 2-4
high potential	K_{oc} = 10,000 to 10,000,000	log K_{oc} = 4-7

Vapor Pressure (P_v): the pressure (in mm Hg) that is exerted by a chemical substance in the vapor phase when that phase is in equilibrium with its solid or liquid form. For solid or liquid organic compounds at room temperature, P_v ranges from 10^{-7} to >300 mm Hg (see Lyman et al., 1982). An approximate indication of relative volatility of organic compounds can be made by dividing this range into three categories:

low volatility	<1 mm Hg
moderate volatility	1-100 mm Hg
high volatility	>100 mm Hg

Water Solubility (S): the maximum amount of a chemical (in mg/L or mol/L) that will dissolve in pure water at a specified temperature, usually 25°C. Log water solubility (log S) values for a variety of organic compounds range from 0 to 9 (based on ppb units) (see Verschueren, 1983; Lyman et al., 1982). An approximate indication of relative solubility can be made by dividing this range into three categories:

low solubility	log S = 0-3	S = <1-1000 ppb (0.001-1 mg/L)
moderate solubility	log S = 3-6	S = 1000 to 1,000,000 ppb (1-1,000 mg/L)
high solubility	log S = 6-9	S = 1,000,000 to 1,000,000,000 ppb (1-1,000 g/L)

Transport and Fate in Water

Transport				Fate				
Solubility (mg/L)	Volatilization (H, $t_{½}$)	Sorption (K_{oc})	Downstream transport[a]	Photolysis ($t_{½}$)	Hydrolysis (k_{OH}, $t_{½}$)	Oxidation ($t_{½}$)	Bio-concentration (BCF)	Bio-degradation ($t_{½}$)
__low __med. __high	__negligible __slow __sign. __rapid	__low __mod. __strong __very strong	__unlikely __likely	__neglig. __slow __mod. __rapid	__nonreact. __low __mod. __high __very high	__neglig. __slow __signif. __rapid	__low __mod. __high	__low __mod. __high
Overall assessment of persistence in water:_____								

[a] Based on residence time or half-life.

Transport and Fate in Soil

Transport				Fate				
Solubility (mg/L)	Volatilization (H, $t_{½}$)	Sorption (K_{oc})	Groundwater transport[a]	Reduction ($t_{½}$)	Oxidation ($t_{½}$)	Plant uptake	Aerobic Biodegrad. (k, $t_{½}$)	Anaerobic Biodegrad. (k, $t_{½}$)
__low __med. __high	__negligible. __slow __signif. __rapid	__low __mod. __strong __very strong	__unlikely __likely	__neglig. __slow __mod. __rapid	__neglig. __slow __mod. __rapid	__low __mod. __high	__neglig. __low __mod. __high	__neglig. __low __mod. __high
Overall assessment of persistence in soil:_____								

[a] Estimated from solubility, sorption and soil equilibrium partitioning data

Transport and Fate in Air							
Transport			Fate			Reactivity	
Vapor Pressure (mm Hg)	Vapor Density (air = 1)	Stratosphere transport[a] (τ)	Photolysis uv absorp.[b] ($t_{½}$)	OH (k_{OH}) ($t_{½}$)	Oxidation (k_{O_3}) ($t_{½}$)	Smog Form.	Ozone Deplet.
__negligible __low __mod. __high	__low __mod. __high	__unlikely __likely	__unlikely __likely	__nonreact. __low __mod. __high __very high	__nonreact. __low __mod. __high __very high	__unlikely __likely	__unlikely __likely
Overall assessment of persistence in air:_____							

[a] Estimated from atmospheric residence time, τ

[b] UV Absorption cutoff 290 nm